

PLANNING DECISION FRAMEWORK FOR BRACKISH WATER AQUIFER,
STORAGE AND RECOVERY (ASR) PROJECTS

By

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A DISSERTATION PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
OF THE REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY

UNIVERSITY OF FLORIDA

2005

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by

Christopher J. Brown

This document is dedicated to my wife, Laura J. Hansen-Brown, who has provided inspiration and support throughout my 4 years at the University of Florida. Without support from Laura Jean, none of this work would have been feasible. In addition, I dedicate this work to my co-workers at the U.S. Army Corps of Engineers in Jacksonville, Florida, who have supported me with flexibility and patience.

ACKNOWLEDGMENTS

The preparation of a dissertation is a long and difficult process. It is only made possible through support from many different persons. I would like to thank my wife, Laura J. Hansen-Brown, for her unwavering support, love, and help through four long years. I would like to thank my family members for their support. I would like to thank my co-workers including Dr. Greg Whittle, Dr. Samir Itani, Mike Fies, Susan Sylvester, Jeff Hendel, Steve Sutterfield, and Rebecca Weiss. I would like to extend thanks to my supervisors at the U.S. Army Corps of Engineers in Jacksonville, including Mr. Luis Ruiz, Mr. Randy Bush, and Mr. Steve Duba. I also am thankful for the technical advice from co-workers and friends at the South Florida Water Management District.

I wish to acknowledge all researchers and scientists from around the world who provided me data and case study information including R. David Pyne, Paul Pavelic, Dr. June Mirecki, Ian Gale, Mark McNeal, Kevin Mooris, Susan Mulligan, Don Kendall, Everitt Wegerif, Rick Nipper, John Zimmerman, Randy Beavers, Ted Corrigan, Paul Eckley, Dave Winship, Billy Smith, Bob Schwartz, Rick Lahy, Dr. Mark Pearce, Larry Eaton, Dr. Jon Arthur, Paul Stanfield, Dr. Mary Anderson, Doug Geller, Joel Hall, Mike Zygnerski, Dr. Tom Missimer, Dr. Christian Langevin, and Dr. Pearce Cheng. I acknowledge a special thanks to my University of Florida advisors including Dr. Kirk Hatfield, Dr. Louis Motz, Dr. Mark Brenner, Dr. William Wise, and Dr. Jon Martin. Drs.

Hatfield and Motz especially deserve singular thanks for the hours of time spent with me helping improve my research and publishing the results.

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Abstract of Dissertation Presented to the Graduate School
of the University of Florida in Partial Fulfillment of the
Requirements for the Degree of Doctor of Philosophy

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December 2005

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Aquifer Storage and Recovery (ASR) projects are being utilized for an ever-increasing number of water management projects nationwide. A large number of ASR projects are in operation in the United States with many others in the early planning stages. The planning of ASR projects involves consideration of numerous qualitative and quantitative factors including site location, applicable regulations, availability of water, type and quantity of water demands, source water quality, subsurface hydrogeology, ambient groundwater quality, geochemistry, geotechnical constraints, and many others. Currently, the planning of potential ASR projects is not standardized. Often, feasibility of ASR is based upon an incomplete consideration of important planning factors. ASR practitioners would be provided a great benefit if a formalized ASR planning decision framework could be developed. This is especially true for ASR sites located within brackish groundwater environments. For these sites, a formalized framework is lacking

that includes reliable planning metrics designed to permit a quick assessment of project feasibility.

The primary purpose of this research project was the development of a formalized planning decision framework for new brackish water ASR projects. The framework consists of planning procedures, qualitative guidance, quantitative guidance, and new ASR performance metrics. The framework is designed to aid the ASR community of practice in assessing the feasibility of prospective projects. The framework, guidance values, and new metrics are based upon a compilation of literature, summaries of 50 different ASR projects from around the world, and extensive numerical modeling of ASR performance. The framework, guidance values, and performance metrics were validated at two ASR project sites not included in the initial data compilation and then was applied to five proposed ASR projects under consideration as part of the Comprehensive Everglades Restoration Plan (CERP).

CHAPTER 1 INTRODUCTION

Description of Artificial Recharge

The concept of underground storage of water has been around for millennia. Nomads in the Middle East have long recognized that sand dunes are a dependable source of recharge after rain events. The nomads of Turkmenistan capitalized on this knowledge by digging long trenches extending radially from various sand dune systems (Pyne, 1995). The Romans utilized cisterns for water storage for generations. The modern term “artificial recharge” has been recognized only recently through efforts of notable engineers and scientists such as Cederstrom, Harpaz, and Bear. Their efforts, and those of many others, have provided the theoretical basis and practical experience to successfully implement artificial recharge projects.

Recharge to groundwater aquifers is ultimately derived from rainfall that falls on the land. Much of the rainfall that occurs on the land runs off to nearby surface water bodies, evaporates, or is abstracted for various water supply uses. Only a fraction of the total rainfall reaches subsurface aquifers via percolation into the ground. Recharge to a surficial water-table (unconfined) aquifer is generally higher than comparable recharge amounts to deeper, confined, aquifers due to proximity to the land surface. According to the American Society of Civil Engineers (ASCE), recharge is

the replenishment of ground water by downward infiltration of water from rainfall, streams, surface depressions, and other sources, or by introduction of water directly into an aquifer through wells, galleries, or other means. Recharge can be either natural or artificial. (ASCE, 2001, p. 3)

Modern artificial recharge (AR) is the process of augmenting natural recharge of groundwater aquifers. According to the National Research Council (NRC), AR is:

a process by which excess surface water is directed into the ground – either by spreading on the surface, by using recharge wells, or by altering natural conditions to increase infiltration – to replenish an aquifer. (NRC, 1994, p. 1)

AR provides a means to store water underground in times of water surplus to meet demand in times of shortage. Water recovered from AR projects can be utilized for a variety of potable and non-potable uses. AR can also be used to control seawater intrusion in coastal aquifers, control land subsidence caused by declining ground water levels, maintain base flow in some streams, and raise water levels to reduce the cost of groundwater pumping (NRC, 1994).

Recharge can be introduced through various surface infiltration methods or through wells. Recharge can be introduced into the saturated or unsaturated portions of an aquifer. Both unconfined and confined aquifers have been used for AR (ASCE, 2001). Surface spreading methods are mainly amenable in unconfined aquifers (Asano, 1985), while wells are utilized to recharge confined aquifers (NRC, 1994).

Surface spreading methods introduce water at the land surface into unconfined aquifers. The percolated water passes through permeable geologic materials and enters into the water table aquifer. In arid regions, a substantial unsaturated zone exists to accommodate surface recharge operations. Usually, a groundwater mound forms beneath the surface spreading basin as infiltrated water reaches the water table aquifer (Abe, 1986). According to Asano (1985), long-term losses from surface spreading are predominantly due to evapo-transpiration. Technical factors affecting feasibility of such an approach include infiltration rates, depth to water table, degree of hydrologic

continuity between surface and water table (Wilson, 1979). Sediment and biological clogging, shallow perched zones controlled by geology, air entrainment, and temperature differences between groundwater and recharge water, and salinity of recharge water control infiltration rates. Surface systems must be tailored to local hydrogeology, quality of input water, and climate (NRC, 1994). Surface spreading can be accomplished through recharge basins, in-channel recharge zones, off-channel recharge zones, leach mounds, and other techniques. Surface spreading techniques are generally easier to develop as compared to recharge wells and both approaches have advantages and disadvantages. According to Abe (1986), advantages of surface spreading include:

- Construction costs are generally much lower for spreading facilities than recharge wells.
- Operation and maintenance costs are generally lower and little to no water treatment is required prior to recharge.
- Large volumes of water may be recharged rapidly.
- Channel modification takes advantage of natural hydrologic connections with unconfined aquifers.
- Recharge near streams may provide for environmental enhancement.

According to Abe (1986), disadvantages of surface spreading include:

- Evaporation losses may be higher than for recharge wells
- Time lag between recharge and recovery may be substantial.
- Land availability or costs may be substantial
- Floods or animals (turtles, rodents, etc.) may cause severe damage to containment dikes periodically
- Surface recharge basins can become an attractive nuisance (e.g., attracting mosquitoes or potential for drowning).
- Periodic scraping may be required in the surface recharge basins to restore full infiltration capacity.

Thick unsaturated zones may be problematic unless a complete subsurface characterization has been completed due to possibility of controlling geological features. These features may lead to a circuitous flow path that may cause excessive time to elapse before recharge water reaches the water table.

If surface spreading methods are infeasible for conducting AR projects, wells may be employed for this purpose. Recharge wells are similar to regular pumping wells. In unconsolidated aquifers consisting of sands, gravels or silts, the wells consist of a well casing, well screen, well filter pack, and an injection pipe. In bedrock aquifers, the recharge well may have an open-hole interval rather than a well screen. Recharge wells located in confined aquifers may have several small diameter recharge pipes terminated in different aquifer zones. Most recharge wells are connected to pumps that allow recharge to occur at high rates. Beyond their increased costs, the major problem with injection wells is clogging of the aquifer around the well, especially at the borehole interface between the filter pack and aquifer where suspended solids can accumulate (NRC, 1994).

Asano (1985) argues that in order to maintain high intake rates for recharge wells, operation and maintenance of the well system must include water treatment prior to recharge and periodic well redevelopment. Sternau (1967) and Wilson (1979) recommend dual-purpose well systems to save the cost of well construction and conveyance systems. They also offer two advantages: namely, installed pumps may be used for well redevelopment and to recover stored water. To maximize recharge and pumping efficiency, the bottom of the recharge conduits should remain submerged. This

also eliminates the problem of air entrainment in the case of water cascading down the well casing. According to Abe (1986), advantages of recharge wells are:

- Recharge wells introduce water directly into the water table or aquifer with minimal losses.
- Recharge by wells may be the only suitable means for recharge due to geology or land restrictions.
- Water recharged by wells can be recovered quickly and efficiently. Slow groundwater movement allows recovery out of the same well usually.
- Recharge wells may reduce conveyance costs.

According to Abe (1986), disadvantages of recharge wells are:

- High construction costs including well and treatment plant construction.
- O&M costs are high primarily due to water treatment prior to recharge.
- In highly permeable aquifers, maintaining a positive outward head is expensive and requires large volumes of water.
- Capacity is much smaller than surface-spreading facilities.

A recent EPA study reviewed 23 categories of Class V injection wells in the United States (EPA, 1999a). EPA estimates that there are 686,000 Class V wells in the USA.

The two largest categories are storm water drainage wells (248,000) and large-capacity septic systems (353,000) that together comprise almost 88% of the totals. The EPA also noted that a few categories like subsidence control wells have less than 100 recorded total wells. Class V wells exist in virtually every state in the USA. Salt-water and subsidence control wells exist in only six or fewer states. The study identified 315 salt-water intrusion barrier wells and 1,185 AR wells out of total Class V wells surveyed.

In a continuation of the Class V injection well study, the United States Environmental Protection Agency (EPA) inventoried 1,185 AR wells within the United States (EPA, 1999b). Approximately 89% of the AR wells are located in 11 states

including California, Colorado, Texas, Florida, Idaho, Nevada, Oklahoma, Oregon, South Carolina, Washington, and Wisconsin. Table 1 provides the well distribution for these states.

Table 1-1. Distribution of 89% of AR wells in the United States

State	Number of AR wells
California	200
Colorado	9
Florida	+/- 488*
Idaho	48
Nevada	110
Oklahoma	44
Oregon	16
South Carolina	55
Texas	67
Washington	12
Wisconsin	2

*Some of the wells may be lake control wells. Source (EPA, 1996b)

AR projects have been tested and successfully utilized by a number of United States government agencies including the Bureau of Reclamation (1996a, 1998), United States Geological Survey (Vecchioli and Ku, 1972; Fitzpatrick, 1986; Buszka et al., 1994), and United States Army Corps of Engineers (Wilson, 1979), in a variety of environments. AR is increasingly utilized by the private sector worldwide to help provide dependable water supply (ASCE, 2001). Outside the United States, AR has been used extensively. Fifteen percent of Germany's water supply needs are provided via AR (Schottler, 1996). In Israel, borehole injection into a dolomitic aquifer has recharged on average 1.1 million cubic meters per year over a 23-year period (Guttman, 1995). South Africa has utilized AR to replenish aquifers for decades (Murray, 2004).

Description of Aquifer, Storage and Recovery (ASR)

ASR is a simple concept in which water is stored in subsurface permeable aquifers when water is plentiful and extracted during times of peak demand. According to the

British Geological Survey (Jones et al. 1999, p. 3), ASR is a sub-set of artificial recharge and is defined as:

storage of treated, potable water in the aquifer local to the borehole(s) that is (are) used for both injection and abstraction. A high percentage of the water injected is abstracted at a later date and the scheme may utilize an aquifer containing poor quality or brackish water, although this does not exclude the use of aquifers containing potable water. ASR schemes enable maximum use to be made of existing licensed resources.

Pyne (1995, p. 6) has defined ASR as

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.

Artificial recharge of groundwater through wells has been explored in diverse settings internationally (Cederstrom 1957; Harpaz 1971; Bichara 1974; Meyer 1989; Bouwer et al., 1990; Johnson and Pyne, 1995; Kendall 1997; Merritt 1997, Peters 1998, Dillon 2002). Groundwater recharge has been utilized in South Florida, New Jersey and California for decades to reverse the effects of salt-water intrusion. ASR wells extend this traditional recharge process by providing the opportunity for subsequent recovery of the water at the same location. ASR wells are a combination of recharge and recovery wells and are commonly designed for seasonal, long-term, or emergency storage (ASCE, 2001).

An ASR site consists of one or more recharge wells, monitoring wells as required, a treatment facility, and a receiving basin or structure for recovered water storage. Figure 1 depicts a typical ASR site. An ASR well is typically a larger diameter production well



Figure 1-1. A typical ASR well (courtesy of CH₂M Hill, Inc.)

with a large wellhead. ASR wells are typically cased down to the depth of the permeable zone used for storage. Monitoring wells are required by permit at most production sites; these wells enable tracking of recharged water during ASR operation. A typical monitoring well is installed 50 to 1,000 feet from the ASR recharge well and has a screen or open-hole interval at a depth equal to the recharge well. In Florida, regulators may also require construction of shallow surficial monitoring wells to detect changes in potentiometric head or discover upward leakage resulting in observable water quality deviations. Treatment requirements vary from site to site with some municipal water utilities using existing treatment capacity during low demand intervals while other ASR users utilize in-line treatment operations on a more regular basis.

The use of ASR is proliferating throughout Florida and internationally with sites in operation or development in the United States, Canada, England, Australia, Israel, South Africa, Tanzania, Netherlands, and Kuwait (Jones et al. 1999). The EPA documented approximately 130 ASR wells within the United States (EPA, 1999b).

ASR technology has been adopted as a groundwater management strategy in many diverse hydrogeologic settings. For example, ASR is utilized in South Florida (confined carbonate aquifer), Virginia and New Jersey (confined clastic aquifers), and Washington (confined fractured-igneous aquifer) for a variety of purposes including urban and agricultural water supply (Brown & Silvey, 1977; Pyne 1995; Reese 2002; Bureau of Reclamation, 1994). However, most ASR systems in the United States are dedicated to recharge and recovery of water that meets potable water quality standards (Pyne, 1995). A few of the systems are very large including the system utilized by the Las Vegas Valley Water District (LVVWD) which consists of 52 wells recharging up to 80 million gallons per day (MGD), (AWWARF, 2003). Increasingly, ASR is conducted using reclaimed storm-water, for subsequent recovery for irrigation purposes (McNeal et al. 1997; Barnett 2000; Gerges et al. 2002). The use of ASR to meet in-stream environmental demands is a relatively new application but one that will be considered in many future projects (USACE, 1999; AWWA, 2002).

Advantages and Disadvantages of ASR

ASR has been recognized as an innovative option for water supply and is being considered in many new projects around the world (Butts, 2002; Durham et al., 2002). ASR has advantages and disadvantages compared to surface storage options (Smith et al., 2000). ASR systems generally require less land acreage, which is particularly suited for urban systems where surface storage is infeasible. ASR systems can reduce water losses due to seepage and evapo-transpiration, which is a key consideration in arid environments. ASR wells can be located in the areas of greatest need, thus reducing water distribution costs, and augmenting surface water supplies (USACE, 1999).

Kasman (1967) noted that using aquifers as cyclic storage reservoirs provided a number of advantages including:

- Lack of permanence
- No loss of storage capacity due to sedimentation
- No loss of water due to evaporation
- Less vulnerability to destruction and contamination
- Absence of downstream threats due to dam failure

Many advantages of ASR wells have been proposed including the following (Pyne 1995):

- Seasonal water storage to meet peak demands
- Long-term storage to meet drought demands
- Emergency source of potable water
- Disinfection by-products reduction
- Restoring natural groundwater levels
- Reducing subsidence
- Improve water quality
- Preventing salt-water intrusion
- Agricultural water supply
- Storage of reclaimed water

Undoubtedly, other applications will be developed also. Many professionals see aquifer recharge as an opportunity to expand water supplies in the future (Bureau of Reclamation, 1994; Stussman, 1997; Durham et al. 2002). Many upcoming applications may not be related to human needs, but they may be developed for in-stream environmental benefits or environmental restoration purposes (AWWA, 2002).

ASR systems can also have significant disadvantages in some cases. Examples of potential disadvantages include low recharge and recovery efficiencies relative to surface storage, which limit capture rates of excess water. Water quality changes that result from mixing with brackish or saline aquifer groundwater may limit the subsequent recovery of stored water (USACE and SFWMD, 1999). Recharge of fresh water into saline aquifers may result in significant buoyancy stratification where fresh water moves upward and out of the permeable injection zone due to density differences (Merritt, 1986; Missimer et al. 2002).

While slightly brackish water may be acceptable for irrigation (less than 1,500 mg/L total dissolved solids) or drinking water (less than 500 mg/L total dissolved solids) purposes, increased salinity, and other water quality changes that result from discharge of ASR water to surface ecosystems, may have unknown ecological effects (NRC, 2002). Detrimental geochemical reactions between the aquifer matrix and the recharged water can increase post-recovery treatment costs, thus rendering a potential project uneconomical (Arthur et al. 2002; Gaus, 2002). Operations and maintenance costs may also be higher for ASR, largely due to high-energy requirements related to pumping or water treatment systems (NRC, 2002a). In some projects, issues involving subsidence can reduce the storage zone permeability or geotechnical integrity of the ASR system (Li and Helm, 2001; Brown et al. 2004). ASR wells can also become clogged due to a combination of biological, chemical, and physical processes (Rinck-Pfeiffer et al. 2000). Well clogging may constitute an ongoing maintenance problem, particularly in ASR systems that manage treated wastewater or raw storm-water.

Regulation of ASR

With the advent of larger ASR well field projects around the world, the regulatory limitations and constraints they present loom large. ASCE noted that legal and regulatory requirements are an important part of any AR project (ASCE, 2001). ASCE recommended the following legal considerations be considered for any project:

- Ability to maintain control of recharged water
- Surface water and ground water storage rights
- Permits and decrees
- Controls on use of reclaimed water
- Liabilities associated with water quality issues
- Type of ownership
- Land ownership
- Site assessments

In the United States, ASR technology often proceeds in advance of regulatory guidelines, so that permit requirements vary from state to state and within a state over time. ASR planners must lobby regulators to develop a more integrated ASR regulatory framework. The regulatory framework provides ASR planners the constraints that potential ASR projects need to work within. In many cases, the existing regulatory framework controls the ultimate feasibility of a promising ASR application. The American Water Works Association (AWWA, 2002) surveyed 46 separate ASR projects in the United States and discussed the overall complexity of the regulatory issues involved in the development and operation of ASR systems. Not surprisingly, the report clearly demonstrates the regulatory inconsistencies within states and across the United

States. The twisted maze of regulations relating to ASR technology may be slowing technological innovation due to perceived risks of ASR implementation.

The development of a new planning decision framework for brackish water ASR projects must include regulatory and legal considerations in its development. A standardized planning decision framework would also enable regulators to “ask the right questions” when evaluating proposed new ASR projects. The use of ASR technology in the United States would benefit from a more holistic regulatory approach that is consistent and based upon a common planning framework. A more consistent regulatory approach to ASR projects across the United States and around the world would enable the technology to be used in more water supply alternatives.

Within the United States, recharge of water through ASR systems is regulated under the Federal Underground Injection Control (UIC) Program. The UIC program was promulgated in 1983 under the Safe Drinking Water Act (SDWA) under 40 CFR 144 and 146.¹ The UIC program and its subsequent amendments were designed to protect current and future underground sources of drinking water. This program is ultimately administered by the U.S. Environmental Protection Agency but management of the program has been delegated to 40 out of the 50 United States (AWWA, 2002), including Florida. The UIC regulations segregate injection wells into five classes from I to V.² ASR wells are regulated as Class V wells and are listed specifically in legislation, as “injection wells not included in Classes I to IV.”³ ASR wells are regulated under this

¹ 42 United States Code 300f and 48 Federal Register 14189, April 1983

² 40 Code of Federal Regulations 146.5

³ 40 Code of Federal Regulations 146.5 (e)

authority. In Florida, ASR wells are defined as “wells not included in the other well classes which generally inject non-hazardous fluid into or above an underground source of drinking water” (Florida Administrative Code, 2003).⁴

A recent study by the U.S. EPA identified 23 categories of Class V wells including septic tank systems, wells that drain storm-water and wastewater, and wells that dispose of mining wastes (EPA, 1999a). The U.S. EPA devoted report volumes to each of the 23 different categories of Class V wells. Volume 21 of the series is dedicated to ASR wells (EPA, 1999b). The U.S. EPA documented 130 ASR wells within the United States (EPA, 1999b). Whereas the other Class V well types generally recharge untreated waters, ASR facilities require treatment of injected water to primary and secondary drinking water standards. Another regulatory impediment for ASR facilities is that treatment may also be required upon subsequent recovery of the recharged waters. This requirement may obligate the owner to invest additional capital to ensure proper disinfection or aeration of the recovered waters. Recovered water-quality criteria depend on its intended use either as a potable supply, or as surface water to meet irrigation or environmental demands. In Florida, the most common use of the recovered water is for potable supply. However, its use for large-scale environmental restoration is under consideration (USACE and SFWMD, 1999).

These regulatory criteria were promulgated to ensure protection of human health and are intended for application at municipal, private or agricultural water plants supplying water to human customers. ASR technology can also be utilized as a water management option to meet in-stream environmental demands. The proposed Everglades

⁴ An underground source of drinking water has a TDS value of less than 10,000 mg/l

Restoration ASR program proposes to use ASR technology in this manner (USACE and SFWMD, 1999). ASR recovered water that is discharged into surface-water for subsequent environmental distribution must conform to both Federal and state regulatory criteria, and the National Pollution Discharge Elimination System (NPDES), both governed by the Clean Water Act of 1970.⁵ These regulations provide numerical definition of inorganic, organic and radionuclide analytes, microbiological pathogens, dissolved oxygen, temperature, and various physical parameters for all waters. Brackish water ASR sites may be hindered by recovered water limitations. In many states, surface-water bodies have been divided among various “classes” of regulation. In the case of Florida, the various classes of surface-water are described in the Florida Administrative Code (Florida Administrative Code, 1996) and include five major sub-divisions as follows:

- Class I – Potable water supply
- Class II – Shellfish propagation and harvesting
- Class III – Recreation, propagation and maintenance of a healthy, well-balanced population of fish and wildlife; predominantly fresh waters
- Class III - Recreation, propagation and maintenance of a healthy, well-balanced population of fish and wildlife; predominantly marine waters
- Class IV – Agricultural water supplies
- Class V – Navigation, utility and industrial use

Most water bodies in Florida have been categorized according to one of the five major sub-divisions. Generally, Class I, II and III provide for the most stringent requirements.

⁵ Public Law 107-303 promulgated in 1970

Texas, California, Florida and Arizona have specific regulations encouraging owners of ASR to reuse wastewater effluent as part of their projects. Arizona allows users of such water to use the water rights to “extinction” (AWWA, 2002). Several states are considering ASR-specific legislation. The state of Washington has published specific rule changes in Chapter 173-157 of the Washington Administrative Code (McChesney, 2001). The rule establishes standards for review of ASR proposals and mitigation of any adverse impacts in the following areas:

- Aquifer vulnerability and hydraulic continuity
- Potential impairment of existing water rights
- Geotechnical impacts and aquifer boundaries and characteristics
- Chemical compatibility of surface and ground waters
- Recharge and recovery treatment requirements
- System operation
- Water rights and ownership of water stored for recovery
- Environmental impacts

Georgia is currently the only state in the United States that has specifically prohibited ASR by statute. Currently, the Georgia Environmental Protection Division (EPD) is completing studies of ASR projects to determine if the ASR development moratorium should be lifted (AWWA, 2002).

Non-traditional water demands include non-contact cooling water or ecological supply. Both of these demand types may require special considerations in order to supply suitable water. Temperature of the supplied water may also be important (EPA, 1977). Extensive work has been conducted to explore the potential harmful effects of temperature changes of natural waters on freshwater fish (Coutant, 1970). Several

western states including Colorado and Oregon have recently promulgated new temperature control regulations to help native fish populations (CWQF, 2003). For all projects, ecological safeguards should be an integral component of any water supply decision process (Zarbock et al. 2001).

Any ASR project of environmental significance may also be subject to requirements under the National Environmental Policy Act (NEPA) of 1970. The NEPA process is utilized to determine if environmental impacts of a proposed project are significant. Projects with significant environmental impacts may involve mitigation of impacts or deferral of the project altogether. Federal water resources projects undertaken by the Bureau of Reclamation, Soil Conservation Service (now the National Resource Conservation Service), Tennessee Valley Authority, or the U.S. Army Corps of Engineers, are subject to NEPA for all projects considered. The U.S. Water Resources Council determined that environmental impacts should be weighed carefully when considering water resources project planning (U.S. Water Resources Council, 1983).

In the future, additional regulation of surface-waters in the United States may lead to ever more stringent discharge requirements. The United States Environmental Protection Agency proposed the development of the Water Quality Criteria and Standards Plan – Priorities for the Future (EPA, 1998). The National surface water quality protection program is a key component of EPA long term planning. This plan describes six new criteria and standards initiatives that EPA and the States will develop over the next decade. The Plan will integrate biocriteria, nutrient criteria, temperature control and microbial pathogen control with improved chemical-specific and whole effluent toxicity criteria into a water quality criteria and standards program that better ensures the

protection of human health and the improvement of surface water quality. The criteria and standards will focus on the following six priority areas:

- developing nutrient criteria and assessment methods to better protect aquatic life and human health;
- developing microbial pathogen criteria to better protect human health during water recreation;
- completing development of bio-criteria as a basis for aquatic life protection (including temperature);
- maintaining and strengthening the existing ambient water quality criteria for waters and sediments;
- evaluating possible criteria initiatives for excessive sedimentation, flow alterations and wildlife; and,
- developing improved water quality modeling tools to better translate water quality standards into implementable control strategies.

It is obvious that many of these focus areas could impact future ASR project development. In the future, it is hoped that regulatory agencies will consider multi-media ASR permits or something similar to simplify the permitting process but still protect human health and the environment. Consolidation of the regulatory framework would lead to more consistent rulings and cost effective water supply solutions (Rogers and Louis, 2002). The issuing of multi-media permits or a consolidated permit for brackish water ASR sites could, in part, be based upon a sound ASR planning decision framework developed as part of this research project.

ASR project regulation has been a growing subject outside of the United States also. Regulatory aspects relating to licensing and charging of ASR schemes are relatively untested in the United Kingdom (Jones et al. 1998). Scientists in the Netherlands, Australia and the Middle East also appear to be grappling with regulatory inconsistencies or misunderstandings (Dillon and Pavelic, 1998). According to Jones, ASR proponents

in the United Kingdom must also contend with European Union regulations regarding recharge water quality, and the potential environmental impact of any proposed scheme (Jones et al. 1999). In Europe, aquifer recharge projects have been conducted for many years, however the experience related to ASR projects is much shorter. There is little experience in using or authorizing ASR projects under existing legislation (Jones et al. 1999).

ASR Use and Constraints

ASR projects are utilized in three broad areas to augment water supplies. The largest and most common use of ASR projects is in support of potable water supply projects. The second most common use of ASR projects is in support of agriculture in the form of irrigation water supply. The newest alternative use for ASR is in support of environmental water supply to support in-stream uses. Each of these categories presents great opportunities to exploit ASR technology, however, each option is subject to many constraints. The primary constraints can be grouped into four general categories including:

- Regulatory (discussed previously)
- Recharge and recovered water quality
- Water availability and demand
- Availability of a suitable storage aquifer

As discussed previously, regulatory thresholds usually dictate limitations on the use of the source water and the recovered water based upon its quality. In addition to the required regulatory thresholds, the end use of the water is of particular importance. Although water may meet all regulatory requirements, recovered water use may be constrained by one or more constituents contained within the water. For instance, recovered water containing moderate amounts of sodium or boron may preclude the use

of the water for irrigating certain crops (Rhoades et al. 1992). Environmental use of the water could be questionable where recovered water contains low concentrations of heavy metals such as selenium or copper. Potable use may be constrained in cases where disinfection by-products form in-situ due to the presence of residual chlorine in the source water. It is evident that water quality constraints may vary for the three primary ASR types. Further information and discussion on the different water quality issues is provided in subsequent parts of this report. All ASR projects are subject to regulatory restrictions under the UIC program and the Clean Water Act. Any ASR use will be constrained by lack of available water or in cases of poor demand. Lastly, the ASR type is irrelevant if a suitable storage aquifer does not exist where excess water can be stored.

Table 1-2 provides a listing of the constraints versus ASR type.

Table 1-2. Primary constraints for different ASR uses

Constraint	Potable ASR Project Type	Agricultural or Irrigation Project Type	Environmental or In-stream Project Type
Regulatory	X	X	X
Suitable Aquifer	X	X	X
Water Availability	X	X	X
Water Demand	X	X	X
Water Quality – Conventional parameters	X	X	Usually not a problem with the exception of temperature
Water Quality – Toxics	X	X	X
Water Quality – Pathogens	X	X	
Water Quality – Disinfection By-products	X	X	X
Water Quality – Emerging Contaminants (e.g., antibiotics, drugs, hormones)*	Unknown	Unknown	X

* A wide variety of these substances have been identified in the world's water.

In summary, in reviewing the four primary constraints to ASR use, water quality is most sensitive to the actual use of the water. The ensuing paragraphs provide a more detailed discussion of the various water quality issues at stake for the three ASR project types. Source water and recovered water quality can be categorized into five areas of interest as noted in Table 1-2 including:

- Conventional parameters
- Toxics
- Pathogens
- Disinfection by-products
- Emerging contaminants

Conventional parameters include total suspended solids (TSS), turbidity, alkalinity, color, chloride, sulfate and total dissolved solids (TDS). These parameters are typically regulated to reduce taste, appearance and odor concerns for potable water users. For ASR systems, they are also important in evaluating potential for well clogging or other operational problems. Besides the actual clogging of the injection well, excess suspended solids or color can lead to changes in water quality precipitated by native aquifer bacteria. In addition, excess color and suspended solids can make other water treatment steps more difficult. For instance, highly colored water may clog sand filters; render ultraviolet (UV) disinfection systems ineffective or lead to the formation of undesirable disinfection by-products. Unfortunately, the use of storm water presents waters with these problematic substances at high concentrations. If the conventional water quality parameters of the source water are manageable, attention must be paid to the various organic and inorganic toxics that may be present. As was reported earlier, the EPA under the SDWA regulates approximately 80 parameters. There are numerous chemical

contaminants that are produced by man-induced activities. Some examples of these include herbicides, pesticides, fertilizers, paints, fuels, etc. Some of these contaminants may end up in storm sewers in urban areas or drainage ditches of large agricultural areas. Many of these anthropogenic compounds are associated with known human health effects. Generally, these are not a concern at ASR sites where the source water is potable water pretreated to meet all primary compounds listed in the SDWA. Toxics may be more problematic if only partially treated storm water or reclaimed water is the ASR source water for storage. Another type of toxic of concern is a toxin created by blue-green algae (Carmichael, 2001). Blue-green algae, also known as cyanobacteria, can form dense growths in mesotrophic, eutrophic, and hypereutrophic waters when the following conditions exist (Carmichael, 2001):

- High nutrients – the cyanobacteria outcompete other algae for available nitrogen and phosphorus
- Temperature – the cyanobacteria have a higher optimum temperature than green algae
- Light – the cyanobacteria grow better at lower light levels than other algae
- Biological – the cyanobacteria have fewer natural enemies, thus lower predation rates, and their ability to float prevents sedimentation

The cyanobacteria can produce taste and odor problems in water as well as natural toxins. The toxins, called cyanotoxins, have been shown to cause acute toxicity and lethality to animals and humans (Carmichael, 2001). Surveys were completed in 1996 to 1998 to determine the presence of the toxin microcystin. Of 677 water samples that were collected at multiple utilities in the United States and Canada, 80% were positive for microcystin (Carmichael, 2001).

Pathogens and disinfection by-products are generally highly inter-related. The term microbe refers to living organisms too small to see with the naked eye. Pathogens are a

type of microbe that may cause disease (Schueler, 1999). Examples of disease bearing pathogens:

- *Shigella* spp. (dysentery)
- *Salmonella* spp. (gastrointestinal illness)
- *Pseudomonas aeruginosa* (swimmers itch)

Some sub-species can cause cholera, typhoid fever and staph infections (Schueler, 1999). The risk of contracting a disease from a pathogen depends upon many factors including:

- Method of exposure
- Pathogen concentration
- Incubation period
- Age and health of person exposed

Protozoa are single-celled organisms that are motile. Examples include *Giardia*, *Lamblia* and *Cryptosporidium*. To infect new hosts, these critters create hard casings known as cysts (*Giardia*) or oocysts (*Crypto*) that are shed in feces, and travel through surface waters in search of a host. The cysts or oocysts are very persistent and can remain viable for many months (Pitt, 1998; Schueler, 1999).

Pathogens are generally a target of water treatment efforts using various methods of disinfection. The most common method of disinfection is the use of chlorination. Chlorine gas is a powerful oxidizer that inactivates the pathogen, rendering it harmless. Unfortunately, a vast majority of water treatment plants within the United States utilize some form of chlorination as the mechanism to achieve disinfection. The chlorination process will create disinfection by-products (DBPs) in water containing color and natural organics (Krasner et al., 1989). The primary families of DBPs produced during chlorination include trihalomethanes (THMs) and haloacetic acids (HAAs). Ozonation is another popular method of disinfection. Ozonation primarily forms only one particular DBP, bromate, however, it can also create recharge water that is super-saturated with

oxygen contributing to possible geochemical reactions in the aquifer. Ultraviolet radiation (UV) can also be utilized for disinfection purposes as long as the UV energy is not absorbed or refracted by highly colored waters (Farrell and Burris, 1995). Other alternative oxidizers include chlorine dioxide and chloramines. All disinfectants and disinfection by-products are regulated by the USEPA under the Stage 1 Disinfectants and Disinfection Byproducts Rule (EPA, 2001).

The National Research Council (1994) cites work done by Bull et al. (1990) to compare the risks from disinfection by-products versus risks from pathogens (which would likely be present without disinfection). Bull et al. (1990) determined that the probability of mortality induced by improperly disinfected drinking water would exceed the carcinogenic risks introduced by DBPs by as much as 1,000-fold.

Besides DBPs, a new water quality concern has arisen related to a class of contaminants called “emerging contaminants”. Emerging contaminants are generally man-made pharmaceutical and chemical compounds including drugs, hormones, antibiotics, caffeine, nicotine, etc. A majority of these substances have been found in wastewater around the world (Montague, 1999). It has long been recognized that sludge from sewage treatment plants contained aspirin, caffeine, and nicotine (Daughton and Ternes, 1999), but it has been work performed by European scientists who have uncovered the real magnitude of the problem (Buser et al. 1998). The effect of these compounds on human health is unknown today. More worrisome is the effect of these compounds on the environment. Environmental users may be more sensitive to low concentrations of certain contaminants than human users.

Potable Use

ASR projects are dominated by potable water supply uses. Potable water projects provide water to both large urban centers (Reese, 2002) as well as rural towns (Murray, 2004). Potable water supply ASR projects typically are co-located with existing water treatment plants (CH₂M Hill, 2002).

Usually, the existing water plants are not fully utilized during off peak times. During off peak times, water demand may be only a fraction of daytime demand. Therefore, water is available for ASR storage during this time. In addition, this water is pre-treated in order to meet all primary and secondary standards under the SDWA. Generally, water quality concerns are focused upon the generation of DBPs from chlorination or ozonation, release of heavy metals from geochemical reactions (Mirecki, 2004), or the potential effects of various emerging contaminants. Although all potable water ASR projects are required to meet regulatory limits for DBPs, residual chlorine dissolved in the ASR recharge water can continue to form DBPs in the storage aquifer (Fram et al. 2002). Recharge water containing high levels of oxygen and dissolved organic carbon can release low concentrations of various heavy metals such as arsenic, manganese, iron, cobalt, nickel, or mercury (Johnson et al. 1998). The water treatment plant associated with the ASR project may not be capable of removing certain heavy metals upon recovery of stored water. Emerging contaminants pose a similar problem.

As the risks from emerging contaminants become better understood, their removal and treatment will become a focal point of research. Some work has already been completed along these lines. Drewes et al. (2002) has shown that removal of anti-epileptic drugs could be accomplished through the use of nanofiltration or reverse osmosis. Raloff (1998) discusses a recent finding that running water through activated-

carbon filters removed all vestiges of the personal care products (PPCPs). Ternes confirmed that activated carbon and ozone removed several PPCPs efficiently (Ternes, 2000).

Agricultural and Irrigation Use

After potable water use, the most common type of ASR project is in support of agricultural irrigation. ASR projects developed in support of agriculture may have less onerous regulatory burdens since water quality restrictions may be lessened (NRC, 1994). Australian researchers have pioneered the use of ASR projects for irrigation purposes (Barry et al. 2002).

The use of AR or ASR wells to supply irrigation water is fairly common in arid regions (Bureau of Reclamation, 1996b). The water quality of the recovered water typically dictates the overall effectiveness of this option. In regions where the ambient aquifer water quality is good, the use of the recovered water is usually not restricted. In the case of brackish water aquifers, the recovered water is a mix of injected and ambient groundwater. Therefore, over time the concentrations of total dissolved solids (TDS) in the recovered water increases until it exceeds accepted regulatory thresholds. The United Nations has conducted significant research in the use of such saline water for irrigation purposes (Rhoades et al. 1992). The principal effects of higher TDS water used for irrigation can range from acute plant stress, decreased crop yield, waterlogged soils, clay swelling, or clay dispersion. Although water containing TDS values as high as 7,000 mg/l have been utilized for irrigation in certain desert areas, the United Nations has focused upon the beneficial use of “slightly saline” water with TDS concentrations ranging from 500 to 1,500 mg/l. Water with those attributes is utilized all over the world (Rhoades et al. 1992). In addition to restrictions related to salinity, certain emerging

contaminants and toxics may also need close scrutiny prior to recommending any ASR project.

An estimated 40% of the antibiotics produced in the United States are fed to livestock as growth enhancers (Raloff, 1998). For ASR projects that utilize storm water, this may be a significant finding and may be an important consideration for system site selection. Manure that contains similar compounds is also routinely spread onto agricultural fields where the compounds can then get washed into surface water or percolate into groundwater (University of Arizona, 2000). Presumably, ASR projects selected for irrigation use will be sited near existing agricultural areas; therefore, emerging contaminants may be present in the source water.

In-stream and Environmental Use

The use of ASR projects to support environmental demands will be an important water supply component in the future. It is already being considered to assist with the restoration of the Everglades ecosystem (USACE and SFWMD, 1999). Unfortunately, the environmental effects of ASR use in this manner are generally unknown. As with the other two ASR types, water quality issues will most likely be the largest constraint to mass use of ASR in in-stream projects. Certainly, ASR could be beneficial in improving water supply, distribution and timing for environmental users. Further research in this area may allow additional projects to be undertaken in this category.

Environmental users may be sensitive to changes in temperature, introduction of toxics, disinfection by-products, and any number of emerging contaminants. Once these contaminants are introduced into surface water or groundwater, environmental receptors may be at risk. For example, nitro musks, used as a fragrance in many cosmetics, have attracted concern because of their persistence and possible adverse environmental

impacts (University of Arizona, 2000). Purdom et al. (1994) documented hormonal effects in fish related to exposure to ethinyloestriadiol (component of contraceptive pills) at concentrations as low as 0.1 ng/l. Studies at Stanford University's Hopkins Marine Station expressed concern about new drugs called efflux-pump inhibitors. Designed to keep microbes from ejecting antibiotics intended to kill them, they also seem to impede the cellular pumps that nearly all animals use to get rid of toxicants (Raloff, 2000). In a recent symposium on the subject, scientists noted that the biggest risks from PPCPs are for aquatic life, not humans (Raloff, 2000). More recent articles echo this thought and recognize the lack of knowledge in this area (Khan and Rorije, 2002). The long-term synergistic effect of these PPCPs on human or ecological health is not known today. Research has been focused upon a few of the most problematic compounds including endocrine disrupters and antibiotics. Endocrine disrupters have the potential to interfere with the production of various hormones that regulate bodily functions. A multitude of endocrine disrupters has been identified in natural waters (Khan and Rorije, 2002). To some scientists the release of antibiotics into natural waterways is even more worrisome (University of Arizona, 2000). The release of antibiotics may result in development of disease resistant bacteria in the various waterways. A number of antibiotics have been positively identified in streams within the United States (Kolpin, 2002).

CHAPTER 2 ASR PLANNING – MEANS AND METHODS

Current Planning Methodology of ASR Projects

ASR project planning consists of multiple parts and iterations. Many planning factors need to be evaluated to determine the ultimate feasibility of a prospective project. As ASR projects are located throughout the world in diverse environments, no two projects are alike. However, many sites share common issues, constraints or problems. A majority of these can be determined through judicious review of the relevant ASR literature including operating site data. Based upon a thorough review of the available literature by the author, no comprehensive ASR site comparison has been completed anywhere in the world. Several investigators (Pyne, 1998; Pavelic, 2002) have compiled information for a limited number of sites to analyze specific ASR issues but no comprehensive evaluation has been completed. Comparisons among brackish water ASR sites are even less available in the literature. A comprehensive comparison of site data would provide an impetus for the development of a new brackish water ASR planning decision framework. According to Grigg (1996), a coordinated framework “provides a structure for the players to work under, a set of process rules, and a reporting procedure”. Grigg goes on to describe additional attributes widely recognized as essential for management actions based upon decision frameworks including the use of effective tools; the basing of actions on scientific and risk assessment; the use of measures to reduce uncertainty; and the selection of financially feasible approaches.

In reviewing existing ASR planning methodologies, one thing becomes abundantly clear; no two sites are treated the same and only limited standardization exists. ASR has been investigated all over the world in all manner of hydrogeologic environments. For most sites, planning has followed recommendations provided in several widely utilized reference books on artificial recharge of groundwater (Asano, 1985; National Research Council, 1994; ASCE, 2001). A limited number of project teams also consulted the only known reference book specifically focused upon ASR projects (Pyne, 1995). In the United States, planning of artificial recharge projects became better-coordinated following passage of the High Plains States Groundwater Demonstration Program Act of 1983.¹ This act directed the Secretary of the Interior, acting through the Bureau of Reclamation and in conjunction with the Administrator of the Environmental Protection Agency, to explore the benefits and impacts of artificial recharge projects in the western United States. Over 13 separate projects have been carried out, with some still ongoing (Bureau of Reclamation, 1996). The various projects utilized available surface water to recharge both unconfined and confined (artesian) aquifers in a variety of hydrogeologic environments using a myriad of artificial recharge techniques. Both surface spreading techniques (Wilson, 1979) and well injection techniques (Bichara, 1974) were tested and monitored. Several of the projects also included recovery of the recharged water similar to ASR projects (Bureau of Reclamation, 1994; Bureau of Reclamation, 1996b). Similar studies were underway in Florida through cooperative efforts by the U.S. Army Corps of Engineers and the United States Geological Survey (Merritt et al. 1983). The initial planning methodologies utilized by the Bureau of Reclamation, United States Geological

¹ Public Law 98-434 promulgated in 1983

Survey and the U.S. Army Corps of Engineers provide a good starting basis for conducting ASR projects. Updating and refining this planning methodology based upon new case-study information would be useful. As an example, the Highline Well Field ASR Demonstration Project (Bureau of Reclamation, 1994) focused planning on the following key issues:

- Water System and availability of water
- Hydrogeologic considerations
- Regulatory considerations
- Water Quality considerations (source water and recovered water)
- Well clogging considerations
- Geotechnical considerations
- Economic evaluation

Of these, geotechnical considerations were only briefly reviewed, while water quality considerations were investigated in a very high level of detail. This level of effort disparity is common among the planning efforts for both the artificial recharge and ASR projects reviewed for this proposal. The investigation effort afforded each planning issue is somewhat dependent upon the agency responsible for performing the investigations. Federal agencies like the Bureau, Corps or USGS, tend to look at all aspects of a problem while private companies may weigh cost and risk issues when determining what to focus upon. Although planning for ASR projects has marginally improved since early efforts, each project is still conducted as a single entity without much regard to what has been successful or what has failed at other projects. Similar to the Seattle project, other recent projects provide a good illustration of the current planning methodology.

The city of Green Bay, Wisconsin had a preliminary ASR feasibility evaluation completed to evaluate the potential for ASR in the region (CH₂M Hill, 1999). The final report reviewed the applicability of ASR in the Wisconsin region. According to the report, ASR applicability is governed by three principal criteria:

Variability must occur in water supply, water demand, or water quality. Planned, useful recovery capacity should be sufficient to justify the initial investment; typically, ultimate capacity should exceed one MGD to be cost effective. A suitable aquifer must be available for water storage. Suitability may be defined in terms of hydraulic characteristics, depth, groundwater quality, geochemistry, competing users, and other criteria. In order to evaluate the three criteria, the following planning factors were investigated:

- Source water supply
- Source water quality
- Water demand
- Geology and hydrogeology
- Other groundwater users
- Geochemistry
- Environmental
- Regulatory
- Economics

An ASR demonstration project in the vicinity of Des Moines, Iowa was conducted in two phases between 1995 and 1998. The first phase was the development of preliminary engineering feasibility evaluation, while the second phase involved demonstration pilot testing of an ASR system. Miller et al. (1998) discusses the project

in detail. The study evaluated similar planning factors discussed previously related to the Green Bay, Wisconsin ASR project. The planning factors investigated included:

- Water supply
- Water demand
- Geology and hydrogeology
- Permitting requirements
- Environmental impacts
- Engineering requirements
- Well performance factors (e.g., hydraulics, well clogging potential)
- Water quality

Anglican Water Services completed a regional feasibility scoping study to evaluate ASR potential in their various service areas (Jones et al. 1999). They investigated limestone, greensand, chalk, sandstone, and sand aquifers. They considered:

- Hydraulic properties
- Structure controls
- Available storage
- Geochemistry
- Availability of recharge water

The study concluded that the Chalk Aquifer and the Lower Greensand Aquifer were the most promising areas. They then evaluated five sites for the Chalk Aquifer using a desktop site selection study. The merits of the alternative sites were evaluated using an analytical decision model. They used several planning criteria:

- Hydrogeology
- Available Infrastructure
- Potential for future ASR development

Jones et al. utilized these criteria to select the Balcerne (Colchester) site and Foxhall (Ipswich) as being most suitable. Unfortunately, desktop studies led to the conclusion that significant additional site infrastructure would be required at each site. Therefore, the existing Horkesley Water Treatment Works was selected for ASR testing in a pilot study.

Murray (2004) lists several AR planning success factors including:

- Quantity, type and reliability of the water source available for recharge
- The quality of recharge water, aquifer geochemistry, the compatibility of the two waters, and aquifer clogging issues – high quality, low turbidity water can be used successfully in any kind of recharge system.
- The hydraulic characteristics of the aquifer and groundwater recovery
- Economics
- Management requirements

Oklahoma State University used multiple planning factors in its evaluation of artificial recharge feasibility in Northwestern Oklahoma (Pettyjohn and White, 1985) including:

- Source of recharge water
- Proximity to source
- Topography
- Permeability of near-surface materials
- Quality of source
- Quality of water in the aquifer
- Availability of source water

Both water quality and source water availability was considered to be of paramount importance similar to the Seattle, Green Bay, and Iowa studies discussed previously. The Oklahoma State study also showed the value of using site selection criteria to develop “overlay” maps of potential recharge sites. Overlay planning methodology is an acceptable system to use for ASR planning or site selection evaluations. Overlay methodology has been utilized by many Federal and State agencies and is discussed in more detail by Focazio et al. (2002).

Both the south Denver Basin aquifer recharge demo project (Bureau of Reclamation, 1997) and the Washoe County, Nevada recharge demo project (Bureau of Reclamation, 1996) included both regulatory and institutional considerations in their planning efforts. The St. Johns River Water Management District utilized both site selection criteria (overlay methodology) and institutional/regulatory considerations in locating new rapid infiltration basins (Rabbani and Munch, 2000). Many of the projects discussed herein also utilized numerical or analytical modeling to aid in the evaluation of project impacts or for estimating benefits. Of the projects reviewed, those that utilized numerical models as planning tools provided the most complete comparison of project benefits and costs.

In planning ASR projects in Portland, Oregon, the Portland Water Works included ASR as part of the Portland area Regional Water Supply Plan so that it could be compared to other water supply options instead of evaluated by itself (Portland Water Works, 2001). The planning efforts reviewed all key project constraints but focused resources upon water quality and effects on the environment because both were key policy priority areas. Whereas the Seattle project viewed ASR by itself, the Portland

approach is more holistic. It regarded ASR as another option for its water supply needs. Any new decision framework should include metrics that allow comparisons of ASR to other more traditional water supply options.

Most of the project planning efforts included some type of pilot study to determine the final efficacy of an artificial recharge or ASR project. Piloting at test sites may be warranted but should be considered carefully. Pyne (1995), Mirecki et al. (1997) and Miller et al. (1998) all relate methodologies and lessons learned through the conduct of pilot tests and the correlation of data to make decisions on larger-scale ASR implementation. Butts (2002, p. 24) notes

any successful long-term ASR project will have had a pilot study of some type performed before final design, construction, and implementation of the full-scale facility.

The South Florida Water Management District and the U.S. Army Corps of Engineers whole-heartedly support the notion of using ASR pilot wells to assist with system scale-up (USACE and SFWMD, 2003). Pilot projects were instrumental in the planning for an ASR project in Chesapeake, Virginia (Dwarkanath and Ibison, 1991). The ASCE Standard Guidelines for Artificial Recharge cautions that ASR pilot project well sites should be selected with care to ensure that results can be extrapolated to the entire recharge area (ASCE, 2001). This cautionary statement is especially important for large ASR well field projects where conditions and planning constraints vary spatially across the project area. For brackish water ASR sites, piloting can be expensive and difficult due to recovery and management of brackish water. In a review of brackish water ASR sites for this report, the author has not uncovered one case of a proposed project that, after initial feasibility studies, was not recommended for pilot testing. Again, this seems to be symptomatic of incomplete feasibility level ASR planning of

these sites. The current planning methodology for brackish water ASR sites seems more experience based rather than following a standardized method.

Planning at Non-brackish Water ASR Sites

In these ASR sites, freshwater from excess storm water or wastewater, is injected into permeable groundwater aquifers for later withdrawal. The aquifers in question may be available for artificial recharge for many different reasons. The aquifers may be over drafted as a result of pumpage in excess of the aquifer safe yield. The aquifer may not be utilized for potable water supply due to objectionable quantities of various constituents such as sulfate, iron, manganese, radon, or others dissolved in the ambient groundwater. Water availability from the aquifer may not be dependable as a water supply due to seasonality of flow or recharge. These aquifers present a tremendous opportunity for artificial recharge or ASR projects. In essence, these aquifers offer an untapped resource that provided the proper conditioning, may provide a cost-effective water resources option.

Planning at Brackish Water ASR Sites

In these ASR sites, freshwater from excess storm water or wastewater, is injected into aquifers with brackish groundwater. According to Drever (1997), brackish waters have total dissolved solids concentrations of between 1,000 to 20,000 mg/l. During aquifer recharge operations, the injected freshwater mixes with the poorer-quality ambient groundwater. Mixing is due to advection, hydrodynamic dispersion, and buoyancy stratification. During recovery periods, the recovered water is a mix of the injected and ambient water. Typically, the recovery of the water is halted once the water quality surpasses a regulated value.

In many cases ASR projects can succeed in storing excess water to meet quantity demands but are somewhat limited due to regulatory limitations or use restrictions. The water that can be provided by an ASR system must provide both the quantity and quality required for a particular use. ASR systems may use either groundwater sources or surface water sources. Frequently, surface water sources offer water quality advantages over groundwater sources (Lucas and McGill, 2002). The quality of the water source can sometimes control the process. For instance, potential well locations in urban areas may be subject to contamination from industrial activities or other anthropogenic sources. Locating ASR wells in rural areas must take into consideration likely non-point source contamination emanating from agricultural areas. Herbicides, pesticides, and fertilizers can all pollute source waters in potential project areas. Although pristine areas may not exist near water sources or system demands, consideration should be given to source water quality. In general, for brackish water ASR sites, a complex interplay between the source water quality and the ambient groundwater quality, controls the project feasibility. Frequently, these types of ASR sites are more challenging to plan, construct and operate. Planning of these sites has been especially unreliable and problematic. Existing sites range from those that are highly successful to those that are questionable both from an economic and an environmental perspective. Planning for these sites is often hindered by the lack of available performance metrics. Additional research is required to develop appropriate planning performance metrics for brackish water ASR sites.

Formulation of an Improved Planning Methodology

Review of ASR Performance Factors

The performance of an ASR system is controlled by a complex interaction among many variables. It has been explored through examination of existing site data, development of physical models, through development of simple analytical models, and through development of numerical models. ASR performance factors can be sub-divided into several categories. First, many ASR performance factors are dependent upon the intrinsic physical properties of the aquifer storage zone selected including hydraulic conductivity, porosity, thickness, heterogeneity, strength of aquifer materials, character of aquifer zone, dispersivity, and tortuosity. In confined aquifer environments, intrinsic physical properties of the confining units may also be relevant performance factors. Another category of ASR performance factors is linked to boundary conditions present at the well site including pre-existing aquifer gradient and density of water in the storage zone. A third category of ASR performance factors is linked to geochemical reactions that may occur in the aquifer after introduction of injected source water. A last category of ASR performance factors is dependent upon site operational considerations including injection rate and volume, quality of the source water, type of source water pre-treatment, storage duration, and well design. The degree of importance of each of the performance factors may be different for aquifers characterized as “non-brackish”² or “brackish”. For instance, in brackish water environments, the density of ambient groundwater in the storage zone may lead to buoyancy stratification of the injected freshwater; the stratification can reduce recoverability of the freshwater.

² Ambient total dissolved solids (TDS) concentration less than 1,000 mg/l

Frequently, a combination of performance factors or their interaction may form the basis for a performance guideline or metric. The development of recommended ASR performance metrics will be based upon an evaluation of ASR performance factors and constraints. Recommended performance guidelines and metrics are discussed in Chapter 3 of this research report.

General discussion of all ASR sites

For all ASR projects, performance factors relating to the aquifer itself are important. The aquifer hydrogeology will control the distribution of the injected water within the aquifer storage zone. The aquifer hydraulic conductivity controls the distribution of water level or potential (pressure) in the aquifer. Aquifers exhibiting low to moderate hydraulic conductivity may result in induced high water levels or aquifer pressure. Inordinately high water levels may result in surface flooding or loss of injected water through surface runoff. Inordinately high aquifer pressures may result in local hydraulic fracturing of the aquifer material itself or surrounding confining units in the case of a confined aquifer (Hubbert and Willis, 1957).

Aquifers exhibiting low porosity or aquifers where fluid flow is concentrated in fractures rather than aquifer pore space, may increase groundwater flow velocity or may induce high rates of diffusion (Gale, 2002; Anderson and Lowry, 2004); either of these phenomena may reduce recoverability of injected water.

The character and structure of an aquifer may lead to heterogeneity of the intrinsic properties. Many aquifers are not true “ideal” porous media and are subject to nonidealities (Alpay, 1972). Examples of nonidealities given by Alpay include:

- Pore size distribution
- Dead-end pore space (disconnected pores)

- Geologic stratification
- Non uniform permeability
- Directional permeability
- Geologic structural framework (e.g., faults, dipping strata)
- Geologic stratigraphic framework (e.g., bar deposits, channel fills, glacial features)

These various factors can also lead to complicated diffusion pathways or enhanced mechanical dispersion (Domenico and Schwartz, 1998). Mechanical dispersion is mixing caused by local variations in velocity. Mechanical dispersion is an “advective” process and is discussed by Anderson (1984) and Sudicky (1986). At field scale, it appears that dispersivities increase with scale up to an asymptotic limit (Gelhar et al., 1992). At any given scale, dispersivities in the longitudinal direction (primary flow direction) can range over two to three orders of magnitude depending on the variability in hydraulic conductivity (Gelhar et al. 1992). Schulze-Makuch (2005) updated Gelhar’s work by compiling a larger dispersivity database from various reports and literature. Schulze-Makuch also proposes a relationship between the geologic medium and hydrodynamic dispersion.

The geotechnical properties of the aquifer material may affect the ASR performance. Aquifers confined by compressible clay units can be subjected to changes in effective stress.³ ASR recovery operations subject the confining clay units to cyclical effective stress changes; decreases in effective stress during ASR injection and increases in effective stress during recovery. Increases in effective stress can negatively affect the

³ Effective stress equals the total stress minus the aquifer pore pressure

clay confining units through subsidence or consolidation (Li and Helm, 2001; Brown et al. 2004).

Performance at ASR sites is often expressed as a percentage of the water injected that is recovered. The most common performance metric for these sites is “recovery efficiency,” a concept first presented in the 1970s (Kimbler et al. 1975). ASR practitioners have utilized the term “recovery efficiency” as the metric of choice to evaluate the feasibility and cost effectiveness of prospective ASR projects. The definition of recovery efficiency utilized by most researchers is the total volume of water recovered up until the applicable regulatory limit, expressed as a percentage of the volume injected. Equation (1) provides a mathematical representation of this concept:

$$(1) \text{ Recovery Efficiency (RE)} = V_r/V_i \times 100\%$$

For instance, an ASR project that injects 1,000,000 gallons of water and then subsequently recovers only 100,000 gallons before regulatory discharge limits have been exceeded has a RE of 10%. In most instances, chloride or TDS is the limiting parameter in potable ASR projects. The use of RE is somewhat limited since it does not allow a determination of the actual mass of injected water that is recovered, especially if both the injection water and ambient groundwater contain the parameter that is monitored.

Pavelic et al. (2002) discusses an alternate terminology using the concept of “recovered mass.” As ASR science advances, the recovered mass approach may become an acceptable alternative metric. As of today, RE is utilized almost exclusively throughout the world.

Streetly (1998) developed a model prototype to simulate various controls on ASR performance in potable aquifers. Streetly used a model configured in radial format

spaced at logarithmic intervals from the ASR well to an outer boundary at 20,000 meters. No density differences between injected and recovered water were applied in the model. He utilized a standard set of basic aquifer parameters with a standard format of five, 30-day cycles of injection/ followed by 30 days recovery. Sensitivity runs were used to research the effect of varying parameters by 0.25, 0.50, two, and four. For the basic set of model simulations, Streetly used the following set of aquifer parameters:

- Transmissivity = 2,153 ft²/day with range from 538 to 8,612
- Aquifer b = 164 with range from 41 to 656 feet
- Porosity = 0.05 with range from 0.013 to 0.20
- Dispersivity = 32.81 feet with range from 8.2 to 131 feet
- Diffusion – small and can be ignored except in dual porosity aquifers
- Pumping Rate = 70,640 ft³/day with range from 17,660 to 282,559 (2.11 MGD)
- Cycle length = 30 days with range from 8 to 120 days
- Rest or storage period between injection/recovery = 0 days to 30 days

Streetly concludes that dispersivity has the most dramatic effect on RE. He concludes that permeability and diffusion have no effect on RE in a homogeneous aquifer. Streetly keenly observes however, that the permeability and the dispersivity are related and often associated with each other. Apart from dispersivity, key parameters were all related to the size of the bubble of water injected; a larger bubble radius is associated with a higher RE.

Streetly also shows that RE improves after each cycle but eventually reaches a state of diminishing returns. For his standard run, RE was estimated at 66% for cycle 1 and 80% at cycle 5. His results demonstrate that thinner aquifers provide better RE performance than thicker aquifers. For $\frac{1}{4}$ of the base storage zone thickness, RE

improves to 73%; for four times the base storage zone thickness, RE declines to 57%.

The numerical results presented by Streetly clearly show that hydrodynamic dispersion and volume injected into the aquifer are the most important variables. When the $\frac{1}{4}$ the base dispersivity was simulated the estimated RE was 77% while for four times the base dispersivity resulted in a RE value of 46%. When $\frac{1}{4}$ the base pump injection rate was simulated for 30 days the RE was predicted to be 57% while for 4 times the base injection rate the RE was estimated to be 73%. When the pumping rate was held constant but the injection period was changed, a similar set of results was revealed. This indicates that volume injected is an important variable.

Anderson and Lowry (2004) performed similar numerical experiments to review the effect of different variables on ASR performance. The performance variations were measured using estimates of RE for each simulation tested. The objectives of the research pursued by Anderson and Lowry were to (1) investigate the hydraulic controlling factors on ASR as they relate to recovery efficiency in selected representative hydrogeologic settings in Wisconsin; (2) develop a methodology using numerical flow and transport models whereby the hydraulics of ASR systems can be simulated.

Anderson and Lowry found that the degree of mixing in the aquifer is a key factor governing ASR performance. Model simulations that included advection only resulted in much higher RE than those simulations with both advection and dispersion processes included. In simulations tested, the RE decreased as the regional pre-existing gradient was increased. A large pre-existing regional gradient tends to carry the injected water away from the ASR well where it may not be fully recovered. Anderson and Lowry note that this issue is even more important when small injection volumes are considered.

Lowry (2004) discusses the results of model simulations using small injection volumes in combination with high regional gradients in detail. Anderson and Lowry also clearly show that RE is inversely proportional to dispersivity; as dispersivity values increase, RE values decrease. In addition to intrinsic aquifer properties, Anderson and Lowry also investigated ASR storage times. Generally, as storage duration increases, RE decreases (Figure 2-1).

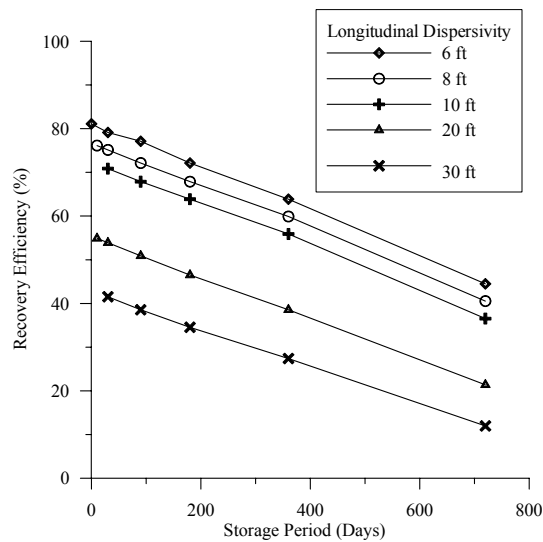


Figure 2-1. The effect of storage period on recovery efficiency [modified from Anderson and Lowry (2004)]

Anderson and Lowry found that RE consistently increased as volume of injected water was increased, except for one case using an unconfined dolomite storage zone, where recovery efficiency decreased after the initial increase. Generally, the RE increases taper off to some asymptotic value dependent upon site-specific conditions at each ASR project site. According to Anderson and Lowry:

these results suggest that pilot tests using small volumes of injected water may not be helpful in estimating recovery efficiency for the larger volumes of water used in final operation of an ASR system. (Anderson and Lowry, 2004, p. 10)

Figure 2-2 depicts the results of model simulations where injection volume was evaluated (Anderson and Lowry, 2004).

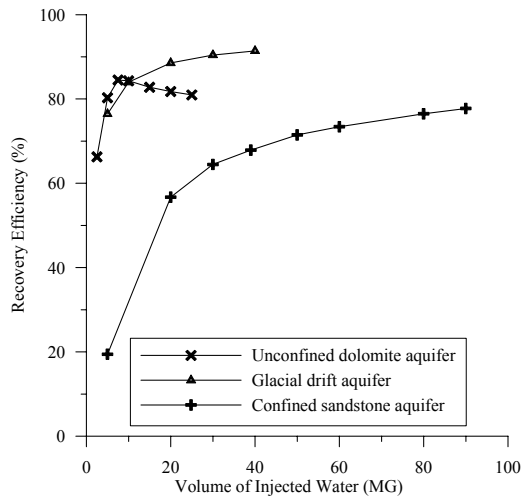


Figure 2-2. The effect of injection volume in million gallons on recovery efficiency [modified from Anderson and Lowry (2004)]

Besides the ASR performance factors related to hydrogeology, ASR performance can also be negatively affected by geochemical reactions that occur between the aquifer matrix and the injected water or the ambient groundwater and the injected water. Gale (2002) noted that geochemical issues were one concern that could constrain ASR development in England. According to the EPA (1999b), the intended use or recharge objective needs to be considered in evaluating quality of water required. The following basic processes must be considered:

- Biodegradation by and growth of microorganisms
- Chemical oxidation or redox
- Sorption and ion exchange
- Filtration
- Chemical precipitation or dilution
- Volatilization or photochemical reactions

Geochemical issues have constrained development or resulted in site closure at several ASR projects in the United States and England. Elevated levels of mercury (Wendell and Glanzman, 1998) and fluoride (Eastwood and Stanfield, 2001) have been noted at two of these sites. Geochemical problems have also been discovered at an ASR site in Green Bay, Wisconsin (CH2M Hill, 2001). Castro (1995) provides a lengthy

discussion of geochemical reactions observed during cycle testing at an ASR site in Myrtle Beach, South Carolina. Arthur et al. (2001) and Arthur et al. (2002) provide extensive discussions of geochemical problems discovered at a few ASR sites in Florida. The report is an evaluation of water-rock interaction at three ASR sites in southwest Florida. The sites include Northwest Hillsborough County Reclaimed Water ASR, Tampa Rome Avenue ASR, and the Punta Gorda ASR facility in Charlotte County. Research focuses upon the latter two sites.

The report is a deliverable from the ongoing “Aquifer Storage and Recovery Geochemistry Project” which has been ongoing since 1998. It was started because preliminary geochemical laboratory leaching experiments conducted in 1995 indicated that uranium and other metals could be leached from Floridan Aquifer System limestones under oxidizing conditions. Source waters with higher dissolved oxygen were noted to be particularly problematic. Arthur et al. (2002), note that a potential metal mobilization problem is indicated when the concentration of a metal is low in both ambient and source waters, but an increase is noted during ASR recovery. The Punta Gorda ASR site clearly shows an increase in arsenic that is above both ambient and source waters. The same pattern is observed at the Tampa Rome Ave ASR facility. Besides arsenic, other metals such as iron, manganese, nickel, vanadium, and uranium are mobilized from the aquifer system matrix into the injected source water and it is contained within recovered ASR water. Arsenic and uranium mobilization are the most consistent and well-documented trends observed in the study by Arthur et al. (2002). Preliminary results indicate that mobilization reactions occur on the order of a few days and can sometimes also be seen in nearby monitoring wells. During three successive tests using similar cycle test

volumes, metal concentrations decrease; however, if larger volumes are introduced that expose virgin aquifer material, additional mobilization would be expected.

Mirecki (2004) studied geochemical data at 11 ASR sites located in south Florida. Mirecki observed that several of the sites had elevated concentrations of arsenic, radionuclides, ammonia, and sulfate. Seven of eleven ASR sites studied reported arsenic results; only one of these sites reported arsenic concentrations in excess of water quality standards. Three of the eleven sites reported ammonia concentrations in the recovered water in excess of Florida surface water regulatory standards.

Jones et al. (1999) discuss geochemical reactions in detail and lists the most prominent ones known:

- Mixing
- Adsorption – onto rock or sediments; metals may have pH dependent properties
- Ion exchange – clay minerals, organic matter, and oxides/hydroxides have capacity for uptake of cations or anions. Brackish aquifers may exchange Ca^{+2} for Na^{+} and result in sodium rich recovered water.
- Oxidation-reduction – Reduction of oxygen, denitrification, reduction of nitrate, oxidation of organic matter, oxidation of sulfide minerals, oxidation of Fe^{+2} leading to precipitation of iron oxides and hydroxides. Oxidation of iron sulfides can induce an increasing acidity, thereby mobilizing metal ions (lead, cadmium).
- Dissolution and precipitation – Mainly determined by the Saturation Index of the mineral in question. If the saturation index is reached precipitation takes place. Precipitation may lead to clogging around the well. Conversely, undersaturated water may lead to dissolution of aquifer material.
- Reaction kinetics could control some of these issues. Kinetics may be slow and may require longer durations to injection/extraction and storage to avail themselves. Longer-term cycles are more likely to reach equilibrium between aquifer mineralogy and recharge water. Minerals like calcite and strontianite dissolve relatively rapidly; and the dissolution of salts like halite and sylvite occur quickly.

Arthur et al. (2002) also proposed the most probable geochemical reaction

mechanisms noted for several Florida ASR sites to be:

- Oxidation of sulfide minerals such as pyrite which may contain trace elements as lattice substitutes (Ni, Co, Cu, Pb, As, Zn, and Mn)
- Desorption or dissolution of Fe and Mn hydroxides
- Oxidation-reduction of organic materials which can mobilize organically complexed As
- Biological transformations

Stuyfzand (1998) has studied similar geochemical issues in the Netherlands. In a study of 11 deep well recharge experiments, oxidation of pyrite is reported resulting in mobilization of arsenic, cobalt, and nickel. According to Stuyfzand, the main variables affecting metal mobility include:

- Native and input water chemistry (Dissolved oxygen, pH)
- Aquifer matrix chemistry/mineralogy
- Input water – matrix contact time and number of cycles
- Site-specific hydrogeology/geochemistry

Gaus (2001) observes that major chemical changes to the recovered water quality are expected when one or more of the following conditions are met:

- There is a large difference in chemical condition between the injected and the native water; this can cause large differences in pH or redox condition
- The native water or sediment do not possess a sufficient pH buffering capacity (e.g., acidic waters)
- There is a large difference in elemental concentrations between the injection and the native water (e.g., fluoride) and significant mixing occurs
- A change in chemical condition of the water having contact with the sediment is able to trigger major (e.g., dissolution of gypsum) or minor (e.g., dissolution of heavy metals) reactions.

Herczeg et al. (2003), discuss the effects of injection of storm water into a brackish limestone aquifer over a five-year period. Herczeg et al. also note that the major effect was observed at a monitoring well located 25 meters from the ASR well. Significant carbonate dissolution (35+/-6 grams of CaCO_3 dissolved per cubic meter of aquifer) and sulfide mineral oxidation was recorded. Less than 0.005% of the total aquifer carbonate

matrix was dissolved during each injection event, and approximately 0.2% of the total reduced sulfur. In addition, measurements of dissolved inorganic carbon values show that substantial aqueous CO₂ was produced by oxidation of organic matter associated with the injected water.

AR or ASR wells may also be subject to corrosion. Corrosion can lead to reduced well efficiency and poor recovered water quality. According to ASCE (2001), corrosion should be a design concern in any of the following cases:

- Low pH acidic water
- DO in water
- Hydrogen sulfide in water
- High TDS (over 500 mg/l)
- Carbon dioxide in excess of 50 mg/l
- Chloride ions in excess of 300 mg/l
- High temp of water in well

Corrosion problems have been reported at one notable project in Texas at the Hueco Bolson AR site in El Paso (Bureau of Reclamation, 1996c). Injection wells have had plugging problems due to improper well development and corrosion. Redevelopment has to occur once every 3 to 4 weeks. Iron, manganese, and zinc compounds have clogged the galvanized steel well screen. The injection wells are subject to electrochemical corrosion. The owners have evaluated several options to alleviate the corrosion problem including applying cathodic protection to the injection wells. Mitigation measures available include corrosion resistant steel, non-metallic casings, greater steel wall thickness (sacrificial), coatings on casing, or cathodic protection systems (ASCE, 2001).

The final general ASR performance category is related to operational considerations of each site. Many operational features are in the purview of the site owner. For

instance, the level of water treatment specified controls the quality of the injected source water. Source water containing large amounts of total suspended solids (TSS) or total organic carbon (TOC) can clog injection wells over time (ASCE, 2001). Well clogging is a serious ASR performance issue that must be considered carefully.

Johnson (1981) reported the principal causes of clogging in an injection well to be:

- Gas binding or air entrainment
- TSS in injection water
- Bacterial contamination of aquifer or filter pack by bacteria growth
- Chemical reactions between recharge and ambient groundwater
- Ion-exchange reactions that could result in clay particle dispersal/swelling
- Precipitation of iron in the injection water as a result of aeration
- Biological changes in the injection water and the groundwater
- Swelling of clay colloids in the dewatered portion of the aquifer
- Mechanical jamming of the aquifer materials caused by particle rearrangement when direction of water movement through the well and aquifer are reversed

Vecchioli and Ku (1972) observed major problems with well clogging during a field AR trial in Long Island, New York. In one test, specific capacity was reduced by half after 10 days of injection. Vecchioli and Ku noted that the Magothy Aquifer (AR storage zone) had a fine gradation and is sensitive to TSS in influent water.

Redevelopment activities resulted in restoration of most of the specific capacity. Usually, the first slug of recovered water from redevelopment activities was very turbid with high concentrations of iron, phosphate, and volatile solids. Vecchioli and Ku also observed that the bacterial content was high in the recovered water indicating some biological clogging in addition to physical clogging by TSS.

Fitzpatrick (1986) noted similar problems at an ASR site in Lee County, Florida. Tests using varying concentrations of influent source water showed severe clogging from TSS and biological growth. Head buildup reached 200 feet in one test. Back flushing helped alleviate this problem. Recovered water contained no coliforms or fecal streptococcus, however, growth of anaerobic bacteria was noted and were determined through observed water quality changes (increases) in ammonia and iron. Decreases in dissolved organic carbon, organic N, and nitrate also occurred.

Bichara (1986) completed extensive laboratory testing on well clogging using different media types and source water with variable TSS concentrations. In general, finer-grained media were the most sensitive to physical clogging by well injection.

Moncaster (2004) provides detailed evaluations of well clogging and efficiency issues for the Portland, Oregon Columbia South Shore well field. Moncaster is still investigating clogging issues in large capacity ASR wells at the Portland location but has attributed some of the problems to the total mass of suspended solids injected rather than just the concentration of TSS in the injected source water.

Injection volume is another performance variable under the control of the owner; larger injection volumes will likely increase the site performance as noted previously in this report. The site infrastructure including the well, pumps and piping, are all controlled through design efforts instituted by the site owner. Proper well design may reduce potential for air entrainment (Pyne, 1995; Groundwater Solutions, 2004). The owner may also be able to elicit some control over the timing of recharge. Source water from lakes or reservoirs may contain high algae counts that can clog injection wells (Bureau of Reclamation 1994; Bureau of Reclamation, 1996b). Temperature of the

source water may affect ASR system performance. Since hydraulic conductivity of the ASR storage zone is a function of density and viscosity, adjustments to the “apparent transmissivity” need to be made to account for lower temperature recharge water (Reeder et al., 1976; Fitzpatrick, 1986; Hickey, 1989; Castro, 1995; Groundwater Solutions, 2004). Lower apparent transmissivity values can lead to higher recorded well pressures during injection operations.

Brackish water ASR sites

The performance of a brackish water ASR system is controlled by complex interactions among many variables. The performance of an ASR system has been explored through examination of existing site data (Harpaz and Bear, 1964; Harpaz, 1971; Reeder et al. 1976; Brown and Silvey, 1977; Pyne, 1998; Miller et al. 1998; Reese, 2002), development of physical models (Kimbler et al., 1975), through development of simple analytical models (Esmail, 1966; Esmail and Kimbler, 1967; Moulder, 1970; Kimbler et al. 1975; USACE, 1979), and through development of numerical models (Khanal, 1980; Merritt et al. 1983; Merritt, 1985; Merritt, 1986; Yobbi, 1996; Merritt, 1997; Huntley and Bottcher, 1997; Gaus et al., 2000; Wright and Barker, 2001; Missimer et al. 2002; Pavelic et al. 2002; USACE and SFWMD, 2004).

ASR researchers in the 1960s (Harpaz and Bear, 1964; Esmail, 1966; Esmail and Kimbler, 1967; Harpaz, 1971) conducted numerous field tests and examined the theoretical basis for storing “freshwater” within saline aquifers. These researchers demonstrated that a number of performance factors were important for storing freshwater within saline aquifers. Among these factors were dispersive mixing, regional groundwater gradient, density gradient, and aquifer material. Harpaz and Bear (1964) noted that long term storage of water in a dolomite aquifer was unsuccessful due to pre-

existing groundwater gradient that caused the injected water to translate down gradient where it could not be recovered. Esmail (1966) introduced the idea of a “density gradient” between the freshwater lens and the ambient saline water within the storage aquifer. He also suggests that a mixing zone resulting from dispersion and diffusion would retard the effect of gravitational segregation, with a smaller rate of “interface laydown” or buoyancy stratification. Harpaz (1971) noted that recovery of freshwater was lower in limestone aquifers as compared to sandstone aquifers.

ASR researchers in the 1970s (Moulder, 1970; Harpaz, 1971; Kimbler et al. 1975; Reeder et al. 1976; Brown and Silvey, 1977; USACE, 1979) were focused upon obtaining results from large-scale field trials and on improving the theoretical understanding of AR. Although the term ASR had not been coined quite yet, the AR field trials in Texas (Moulder, 1970), Minnesota (Reeder et al. 1976), and Virginia (Brown and Silvey, 1977) all provided valuable information on recharge and possible recovery of stored water.

Kimbler et al. (1975) provides a detailed evaluation of storing freshwater in saline aquifers. Both analytical and physical modeling was performed. As part of the theoretical development, Kimbler et al. assumed that:

- Aquifer is horizontal, homogeneous, and isotropic, and is of infinite areal extent.
- Viscosities of injected and native fluids are the same

Based upon both analytical solutions and physical modeling using “mini-aquifers”, Kimbler et al. developed a good understanding of some of the key factors in ASR performance in saline aquifers. Kimbler noted the following important factors:

- Molecular diffusion and convective dispersion
- Segregation of two fluids due to density differences
- Pre-existing groundwater gradient
- Aquifer dip
- Aquifer storage zone thickness

Kimbler also discusses the recovery of the injected freshwater defining a term called “recovery efficiency” (RE). This is the percentage of the volume of freshwater recovered versus the volume injected. Kimbler uses recovery efficiency to examine performance of individual cycles of injection and recovery. Kimbler concludes that RE increases in every cycle if storage times are short and recovery does not remove the entire injected residual. For multiple cycles, he uses the term “cumulative recovery efficiency” (CRE) to account for unrecoverable water that forms a residual in the aquifer. Using the physical models developed for the study, Kimbler et al. determined that thinner storage zones result in higher RE values than comparable thick storage zones. RE of dipping aquifers (greater than 30 degrees) is generally less than comparable horizontal aquifers. Lastly, Kimbler et al. note that the higher the density differential between the injected water and the ambient saline groundwater, the poorer the RE performance. The same is true for a high degree of mixing or high pre-existing gradient.

Reeder et al. (1976) describes a case study in Minnesota where over two million seven hundred thousand gallons of highly treated wastewater were injected into a solution-riddled dolomite storage zone. During the recharge event the temperature of the recharge water dropped from 15 to 11.2 degrees Celsius; a concomitant drop in recharge rate was noticed at the same time (rate dropped from 108 to 90 gallons per minute). In addition, it was noted that although the recharge water contained concentrations of total dissolved solids (TDS) in excess of 3 mg/l, no significant well clogging was reported. Lastly, the mixing of the recharged wastewater was characterized as moderate to high due to aquifer dispersion. Dispersivity of the dolomite zone was determined from a single well tracer test to be approximately 280 feet.

Brown and Silvey (1977) provide a thorough study of AR and recovery of the recharge water at a site in Norfolk, Virginia. The Norfolk field trial was hampered by dispersion of interstitial clay particles contained within the heterogeneous aquifer storage zone. If not for the clay dispersion issues, the Norfolk field trial would have been a highly successful example of freshwater storage within a slightly saline aquifer. The authors estimated that after a few cycles, up to 85% of the recharged water could be recovered prior to exceeding regulatory limits for chloride in drinking water. The predicted performance was attributed to the ambient aquifer water quality (only slightly brackish) and the low amount of mixing observed in the sandy portions of the aquifer.

ASR researchers in the 1980s (Khanal, 1980; Merritt et al. 1983; Merritt, 1985; Merritt, 1986) were focused upon obtaining results from extensive numerical modeling simulations.

Khanal (1980) utilizes a digital model, based upon analytical work completed by Kimbler et al. (1975), to examine controls upon RE performance for ASR wells located in saline aquifer environments. The objectives of the study were:

(1) to predict recovery efficiency of the cyclic storage/retrieval system via mathematical modeling.

(2) to perform sensitivity analyses of the model parameters in order to predict the effect of changing these.

Khanal examines RE based upon data from an existing St. Lucie County, Florida ASR well. Khanal examines the effect of multiple variables upon RE. He reviews the effect of density differential, aquifer storage zone thickness, aquifer transmissivity, and dispersivity. His findings agree with Kimbler et al. (1975) for the most part. Large

density differences between recharged water and ambient saline groundwater (e.g. freshwater versus seawater) lead to poor RE while thin storage zones result in better RE values. High transmissivity values also revealed poor performance with RE declining as transmissivity values increased. Studies of dispersivity were not at the proper scale and the results are deemed inconclusive. Further evaluations determined that the RE value typically declined with long storage times prior to recovery of the injected water. RE values were halved when storage time of 2 years was simulated compared to 180 days of storage. Although it is not discussed in detail within the report, the assigned aquifer gradient is most likely responsible for the observed decrease in RE with higher values of storage time. As was noted by Harpaz and Bear (1964), pre-existing gradients within the aquifer tend to cause the injected freshwater to translate down gradient where portions of it become unrecoverable.

Merritt et al. (1983) discusses results from three field trials conducted in Florida. First, a field trial conducted at the Cocoa Beach water plant near Orlando, Florida, is discussed. Table 2-1 provides key trial data:

Table 2-1. Data from Cocoa Beach ASR field trial

Cycle	1	2	3	4	5
Volume In (Mgallons)	11.2	11	11.6	33	28.5
Storage Time (days)	1	1	1	1	1
Volume out (Mgallons)	3.8	5.7	7.7	14.4	21.4
RE (%)	33.9	51.8	66.4	43.5	75.1
Qin (gpm)	700	700	675	740	665
Qout (gpm)	915	955	985	970	970

The Cocoa Beach trial took place from 1969 to 1970 and five full cycles of recharge and recovery were completed. The storage zone was within the Floridan Aquifer System

and consisted of sandy limestone with an open hole interval of approximately 205 feet. The bottom five feet of the storage zone was identified as “cavernous”. These data clearly shows that RE does improve over time if the storage interval duration is short.

Merritt et al. (1983) also discusses tests in Pinellas County, Florida by Black, Crow, and Eidsness, Inc., 1974. The storage zone was 40 feet thick and contained ocean water. Initial test cycles used short duration storage times (practically zero) so RE was 43% to 46%. The third test cycle utilized a 10 hour storage period resulting in a reduced RE of 39%. Fifth and Sixth cycles with storage periods of at least 16 hours resulted in RE of 1% for each cycle. The poor performance was attributed to buoyancy stratification that happened in the storage zone due to ineffective geologic confinement. Based upon the field results, the authors speculated that very high hydrodynamic dispersion occurred in the Karst-like dolomitic zones located throughout the storage zone.

Merritt et al. (1983) also summarizes work by Wedderburn and Knapp (1983) that was completed at an ASR test site near Jupiter, Florida in 1975 to 1976. The initial cycle test resulted in RE of approximately zero but increased to 35% by cycle 4. Due to the low salinity (chloride concentrations of 2,000 mg/l) value in the aquifer storage zone, the authors speculate that an appreciable degree of buoyancy stratification did not occur. A storage period of 120 days during cycle 4 did not cause a large reduction in RE as was seen in the Pinellas County, Florida site.

The report also observes that RE can be reduced if changes occur in the vertical or lateral distribution of permeability due to plugging of the formation around the well during injection. This phenomenon may result in unrecoverable water becoming trapped in the aquifer storage zone. Lastly, the report explains that phenomena such as (1) degree

of uniform flow, (2) degree of anisotropy in the aquifer and (3) degree of hydrodynamic dispersion are all highly dependent upon the structure of the aquifer material.

Merritt (1985, 1986) used a numerical prototype model to evaluate ASR performance factors. Merritt focuses his efforts on the various factors affecting ASR recovery efficiency (RE). The prototype model was assigned radial hydraulic conductivity (Kh) values ranging from 0.18 to 520 feet per day and vertical hydraulic conductivity (Kv) values ranging from 0.10 to 40 feet per day. Model runs to test buoyancy stratification effects at the initial values utilized (ambient chloride concentration = 2,000 mg/l), revealed that significant buoyancy stratification did not occur and recoverability was not affected. Significant effect was noticed when Kh and Kv were multiplied by at least a factor of 10. When seawater concentrations and densities were simulated in the aquifer (TDS = 35,000 mg/l), more significant buoyancy stratification was demonstrated. RE was reduced in half as compared to the original Kh and Kv values. When Kh was multiplied by 5 times, the RE was reduced to 9 % from an original value of 67 %. In a test with Kh equal to 10,000 feet per day and Kv equal to 10 feet per day, RE was estimated to be 26.7%. Merritt also evaluated the effect of aquifer storage zone thickness upon RE. Merritt's findings echo earlier researchers in that thinner storage zones generally result in better RE performance. Model simulations revealed that RE ranged from 67.5% at thickness of 10 feet to 61% at 50 feet.

Merritt also reviewed the effect of aquifer anisotropy upon RE. The simulations produced mixed results when comparing isotropic and anisotropic aquifers. At higher dispersivity values, RE was almost equal under either case. At low dispersivity values, RE was lower in anisotropic aquifers. Merritt also concludes that the degree of

hydrodynamic dispersion is one of the main controls on ASR performance in saline or brackish water aquifers. Using model simulations with an ambient chloride concentration of 2,000 mg/l and density equal to 62.57 pounds per cubic foot, the RE varied from 28% for moderate dispersion using a dispersivity value of 30 feet to an RE value 74% for a case of low dispersion using a dispersivity value of 2 feet. Merritt (1985) noted that RE was somewhat sensitive to porosity changes and observed that increasing porosity from 20% to 50% led to reduction of RE (from 24% to 14%) for higher dispersivity values with only minor reductions of RE at low dispersivities (from 71% to 64%). In later studies, Merritt (1986) concludes that RE is not sensitive to changes in porosity as long as the values are in a reasonable range (e.g., 30 to 50%).

In the case where recovery of freshwater occurs immediately after injection, RE values were unchanged when including a realistic regional gradient as part of the simulation (e.g., a gradient similar to observed values within the FAS in Florida). When storage periods intervened, RE values dropped. Based upon results of numerous simulations, Merritt demonstrates that the effects of storage time were minor at 6 months but significant at 5 years. Initial RE was 60.9% using 2,000 mg/l of chloride in the ambient groundwater storage zone. After 6 months of ASR storage RE was 59.9%, a slight reduction from the zero storage simulation. After 5 years, RE was reduced to 21.9% due to pre-existing gradient applied in the numerical model.

In his later research efforts, Merritt found that unequal injection or extraction rates did not affect RE. Wells that were not fully penetrating showed very minor changes in RE with reductions of up to 1%, as compared to fully penetrating wells. Merritt did confirm that volume injected was strongly related to RE. Higher Volumes lead to better

RE up to an asymptotic limit. Merritt (1986) provides a figure compiling simulation results for different initial injection volumes. At an injection volume of approximately six million cubic feet of freshwater, the improvements become very small and an asymptote is observed. Similar to other researcher findings, Merritt also shows that RE improves with successive cycles as long as the cycles stop recovery at 250 mg/l Chloride.⁴ For the simulation case with a low dispersivity value of two to four feet with no interlayer dispersion, RE reached a maximum with an asymptote after four to five cycles. At higher dispersivities (e.g., 30 feet), 8 to 10 cycles were required to reach the maximum asymptote. Figure 2-3 depicts the simulation results from Merritt (1986) along with data from the USGS Cocoa Beach ASR trial from 1970. Note that both the simulations and the real field data show that the RE improves from cycle to cycle as long as the storage period is short. The figure also indicates that the number of cycles required to reach a maximum asymptote increases as the dispersivity decreases.

ASR researchers in the 1990s (Yobbi, 1996; Merritt, 1997; Huntley and Bottcher, 1997) were focused upon obtaining results from extensive numerical modeling simulations.

Yobbi utilized the finite-element model, HST3D, to evaluate the importance of various parameters on the recovery of effluent in a saline aquifer (Yobbi, 1996). Yobbi performed model simulations based upon earlier work presented by Hickey and Ehrlich (1984).

⁴ Regulatory limit for drinking water under the Safe Drinking Water Act

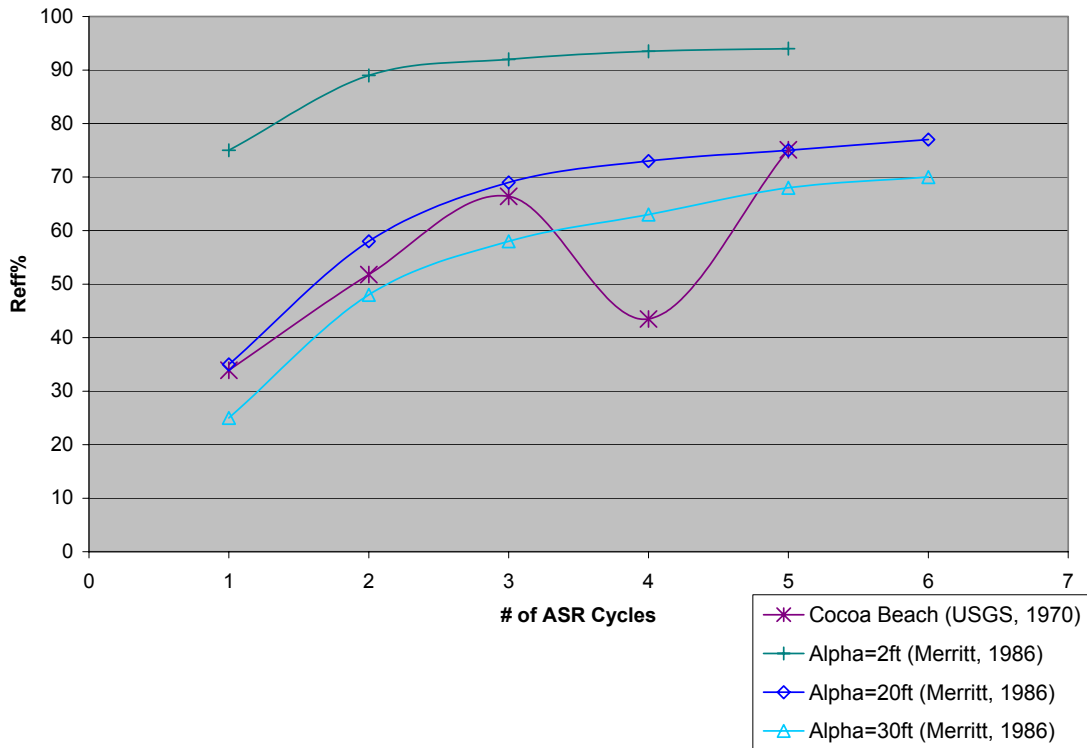


Figure 2-3. Results from historical reports by Merritt et al. (1983) and Merritt (1986)
The y-axis is RE percentage and the x-axis is ASR cycle number.

As the research is compiled and reviewed, similar conclusions are drawn by many of the key investigators. Like earlier researchers, Yobbi determined the following:

- The greater the density difference between the injected effluent and ambient groundwater, the lower the recovery efficiency
- Higher groundwater salinity makes density stratification more likely
- Recovery efficiency decreases markedly as dispersivity increases
- Generally, high formation permeability causes poor recovery efficiencies
- Partial injection well aquifer penetration as compared to complete aquifer penetration has an insignificant effect on recovery efficiency
- Porosity and anisotropy variations do not significantly alter recovery efficiencies

Merritt (1997) provides additional discussion of ASR recovery efficiency in brackish water aquifers. Merritt develops a prototype model of the Hialeah ASR site near Miami, Florida. He constructed a three dimensional model representing the heterogeneous and anisotropic aquifer found at the Hialeah location. The model

simulates ASR performance within the FAS in south Florida. Most of the work is similar to his previous efforts discussed above. However, Merritt does mention the potential for interlayer dispersion in real aquifer systems and how some existing models are not able to accurately simulate this phenomenon. Generally, interlayer dispersion can reduce ASR RE in some cases.

Huntley and Bottcher (1997) developed numerical simulations of a highly layered aquifer system and found reductions in estimated RE as compared to isotropic homogeneous cases. Layered aquifer systems are probably more common than homogeneous systems due to geologic processes such as sedimentation.

ASR researchers in the present decade (Gaus et al. 2000; Wright and Barker, 2001; Missimer et al. 2002; Pavelic et al. 2002; USACE and SFWMD, 2004) were focused upon obtaining results from extensive numerical modeling simulations.

Gaus et al. (2000) describes the progress made in developing models simulating both the physical and geochemical aspects of ASR schemes. SWIFT is a fully transient 3-D model that simulates flow and transport of fluids. It is ideally suited to modeling ASR and has been used to simulate the response of the four major aquifers in the UK – Chalk, Lincolnshire Limestone, Sherwood Sandstone, and Lower Greensand. The aquifers encompass dual-porosity, fractured, and porous-media aquifers.

Sensitivity analyses of the model's responses to matrix porosity, fracture porosity, fracture permeability and thickness were carried out during the investigation. Results showed low sensitivity to parameters in the estimated ranges with only matrix porosity having a significant effect in the Chalk, the Lincolnshire Limestone and the Sherwood

Sandstone. The most important conclusion from the evaluation of ASR performance is that dual-porosity aquifers will need more conditioning than single porosity aquifers.

Wright and Barker (2001) studied the dual-porosity Chalk aquifer in England using a semi-analytical model. The study noted the importance of characterizing the aquifer thoroughly in order to more accurately evaluate potential ASR performance.

Missimer et al. (2002) discusses success or failure of ASR systems based upon local hydrogeology. Predominant factors discussed are aquifer hydraulic properties and density contrast between the water in the storage zone and the injected water. The article states that

if the hydraulic properties are not compatible with the ASR concept being designed, then the system will likely fail regardless of the injection and recovery rate. (Missimer et al. 2002, p. 33)

Most ASR systems that have achieved a high RE occur in slightly brackish-water aquifers or in aquifers with a high degree of confinement. Missimer et al. (2002) reveal that the selection of the right portion of the aquifer storage zone is a key component to a successful project. Based upon model results completed using SEAWAT (Guo and Langevin, 2002), Missimer et al. evaluated potential operational scenarios for the proposed Everglades ASR project (USACE and SFWMD, 1999). The study noted that while 80% RE could be achieved within five years in moderately brackish storage zones (4,000 to 5,000 mg/l TDS), an economical RE could not be reached in 25 years for storage zones containing seawater. The major conclusion was that the ASR project would fail under these circumstances because of the density differences.

Missimer et al. also evaluated highly permeable storage zones with moderately brackish ambient groundwater and determined that the ASR system would also fail or have extremely low RE values. The authors recommend an aquifer transmissivity

ranging from 6,700 ft²/day to 47,000 ft²/day and ambient groundwater with a TDS value of less than 20,000 mg/l. Due to possible density stratification, the aquifer should also be well confined on top of the storage zone with some basal confinement below the storage zone. Missimer also studied the effects of large-scale anisotropy. It was noted that anisotropy greatly influences the distribution of injected water and possibly “channels” the stored water in certain areas where the water may become unrecoverable. In stratified sedimentary rocks, there is a disparity between the hydraulic conductivity in the horizontal plane versus the vertical plane. Common ratios of horizontal hydraulic conductivity to vertical hydraulic conductivity range from 5 to 100. Stratification plays an important role in defining the geometry of subsurface injected freshwater. As the stratification of the aquifer becomes more pronounced, buoyancy stratification of the injected freshwater becomes less severe. In cases where formation stratification or vertical anisotropy is negligible, buoyancy stratification of injected freshwater can be quite problematic due to density differentials. Due to buoyancy stratification, injected freshwater moves upward to form a cone-shaped plume. While in the anisotropic aquifer, the upward movement of injected water is restricted, so that a ball-shaped plume or bubble is formed. Injected freshwater can also follow preferential flow pathways creating poor RE due to “fingering flow” where dense water sinks into freshwater that has been injected.

Pavelic et al. (2002) used a combination of numerical model simulations and dimensionless parameters to measure the effectiveness of various contrived ASR systems. Pavelic et al. identified the two key performance factors that affect ASR RE, namely dispersion associated with aquifer heterogeneity and displacement of injected

freshwater due to regional groundwater flow. As part of the study, Pavelic et al. devised two simple dimensionless lumped parameters to evaluate each of the two key ASR performance factors. The dimensions of the injected water plume may be estimated by determining the radius of an idealized cylinder of water within the aquifer storage zone. This idealized radius assumes that no dispersion takes place during the injection period. Therefore, this value termed r_m by Pavelic et al. can represent an advective plume of water. The amount of dispersion is generally controlled by the intrinsic dispersivity (α). Pavelic et al. proposed using these two terms in a ratio to create a dimensionless parameter termed “relative dispersivity”. The relative dispersivity is a ratio of dispersivity to r_m or α/r_m . As this dimensionless ratio gets larger, ASR RE decreases since the amount of dispersion grows larger compared to the advection. Pavelic et al. also propose a second ratio called the “relative drift” which is the ratio of the effective displacement of the injected water (due to pre-existing gradient) to r_m . Pavelic et al. use the proposed performance ratios in combination with numerical modeling to determine some additional ASR performance considerations. First, similar to other researchers, larger injection volumes can lead to higher values of RE. Second, storage times greater than 100 days combined with higher regional gradients tend to reduce the RE. As dispersion increases, RE decreases.

The USACE and SFWMD (2004) performed numerical modeling of proposed ASR projects in south Florida. The modeling determined that RE is also sensitive to the regulatory regime. For instance, for a potable ASR project, under Federal drinking water standards, recovery of freshwater could continue until the chloride concentration exceeded 250 mg/l, while for an environmental ASR project, under State of Florida

surface water regulations, recovery duration is governed by the concentration of specific conductance. Through model simulations, the USACE and SFWMD found that at the Hillsboro ASR pilot project site, RE would be better if the recovered water was proposed for use for potable purposes as compared with its intended use for environmental users. This finding is significant and reinforces the concept of optimal ASR well site selection.

For brackish water ASR sites, performance metrics are generally lacking. For ASR projects utilized in support of irrigation, additional parameters of interest would be sodium, boron, and herbicides. For ASR projects supporting environmental in-stream uses, critical parameters of interest are unknown at this time. It is easy to see that recovery efficiency by itself is not a sufficient performance metric since it is highly dependent on the use and regulation of the recovered water.

Review of Other ASR Site Selection Factors

Typically, ASR sites are also located to maximize their effectiveness and minimize potential environmental impacts. Various site selection factors may be important including location of:

- Existing groundwater users
- Important habitat
- Reliable power
- Available source water
- Customers or water users
- Suitable hydrogeology

Collection of Existing ASR Site Data

The formulation of an improved planning methodology for ASR projects should be based upon results and “lessons learned” at existing operating projects. With that theme in mind, the author undertook a large data collection effort for this research report. First, existing published data were collected and collated. Published ASR data ranged from a

brief project summary to extensive reports. Some published data were available for a total of 30 sites. In addition to the published data, the author contacted numerous ASR owners and developers across the USA, Australia and England to request key ASR operating data. A total of 20 ASR project sites were contacted and agreed to send data. The data sent by the ASR proponents varied in importance and scale as well as format (e.g., hard copies vs. electronic data deliverables). A net sum of 50 sites is discussed herein and located on Figure 2-4.

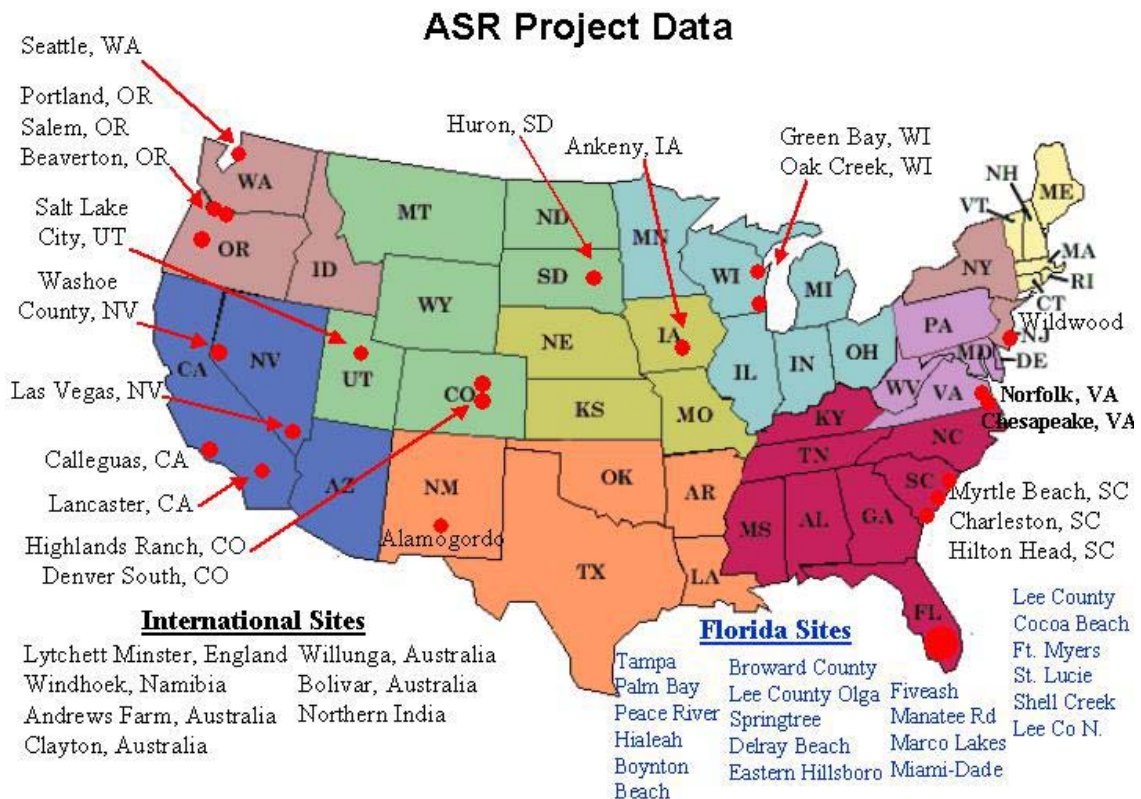


Figure 2-4. Location of ASR project sites where data were collected for this report.

The available data were organized into several categories. First, basic site background information was gleaned from the datasets. Relevant basic site data consists of the site location, geologic environment, and ambient groundwater quality (brackish water vs. freshwater). For approximately one third to one half of the ASR project sites,

operational data included influent level of total suspended solids (TSS), degree of well clogging observed, extent of disinfection by-products recorded at the site, extent of geochemical issues, and total cost per 1,000 gallons to develop the water supply. The various data collected from each of the 50 sites was then compared and contrasted. Lastly, key findings regarding the 50 sites were developed with an emphasis towards improving operation at future projects such as the Everglades ASR program (USACE and SFWMD, 2004). For this research effort, brackish water ASR sites have been segregated from non-brackish water sites since each is significantly different.

Non-brackish water ASR sites

The non-brackish water ASR sites reviewed for this report include sites across the United States, England, and Namibia. They represent diverse geologic environments as well as different operating types. Twenty sites were reviewed for this effort. Available data ranged from extensive reporting for Oak Creek and Green Bay, Wisconsin to short summaries available for the Huron, South Dakota site. Electronic data were provided for a number of sites also. This data facilitated development of unit water costs. Figure 2-5 shows the location of the non-brackish water ASR sites utilized in this report.

A brief synopsis of each site is provided in this report to aid with the development of an ASR planning decision framework. The brief summary will detail basic site information (if available) along with a discussion of special issues or “lessons learned” from the site. These sites were also selected due to the wealth of data available for each one in the literature. In addition, for several of the sites, the author secured additional reports and data that have not been widely published. Generally, more data were available for these sites than were readily available for the brackish-water project sites.

For that reason, discussions of some of the brackish-water sites available later on in this report are abbreviated.

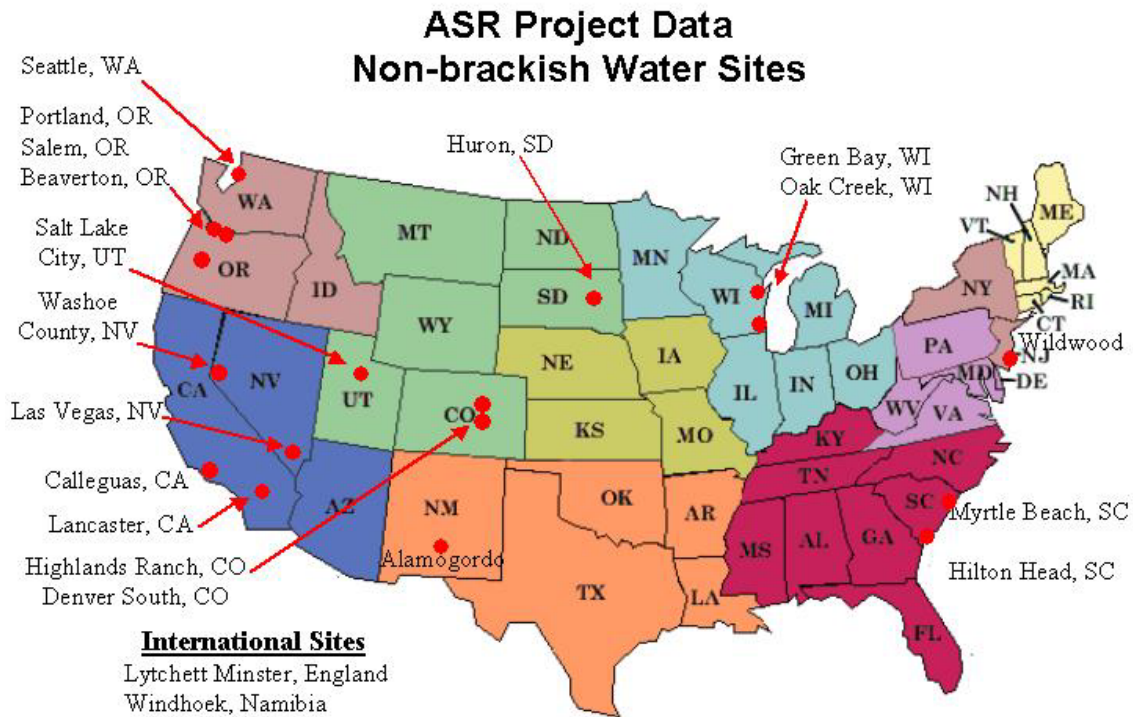


Figure 2-5. Location of non-brackish water ASR project sites where data were collected for this report.

Seattle ASR project. The Seattle, Washington ASR site, also referred to as the Highline well field site, is located immediately south of the city downtown. The Highline well field was designed to provide supplemental water supply to the city, augmenting the supply from Lake Youngs, a reservoir located southeast of the city. Groundwater modeling of the system revealed that peak demand pumpage of 6 to 8 MGD would result in significant long-term groundwater declines (Bureau of Reclamation, 1994). A solution to this problem was to utilize ASR technology to provide aquifer recharge when excess

surface water was available. The ASR system was constructed and tested in 1991. Full-scale single-well and multi-well testing was performed from 1991 to 1993.

The well system is screened in unconsolidated glacial deposits characterized as highly permeable sand, gravel, and cobbles. The aquifer system has an average transmissivity of 3,000 ft²/day and is a confined to semi-confined aquifer.

Water quality in both the aquifer and source water was deemed to be acceptable with minor amounts of radon, manganese, iron, and zinc present. Total suspended solids of the treated source water ranged from 1.0 to 4.0 mg/l. Recharge operations were delayed twice during the project due to flooding impacts in late 1990 (e.g., associated highly turbid source water) and in 1992 due to drought conditions that affected source water flows and permissible diversions. In addition, during summer months, excessive algae growth in Lake Youngs resulted in moderate clogging of the ASR well. This clogging was found to be reversible through periodic back flushing or well re-development. During recovery operations, minor amounts of disinfection by-products (total trihalomethanes) were observed but at concentrations well below regulatory levels. Minor geochemical enrichment of manganese, iron, and radon was also noted during ASR recovery operations. Radon levels frequently exceeded proposed regulatory standards at that time (e.g., greater than 300 pCi/L). Site engineers noted that aeration of the water could remove the radon from the recovered water if water blending was not possible. Recovery of the injected water was highly successful during pilot testing operations. After several cycles the recovery efficiency approached 100%. Geotechnical problems resulting from ASR operation were explored at the site but found to be minimal. Subsidence of overlying clay confining units was not deemed a problem due to

past geologic history and pre-consolidation from the weight of glacial ice. The Seattle Water Department completed a thorough cost analysis of the project and determined that the overall unit cost was approximately \$0.54 per 1,000 gallons recovered. This unit cost was significantly less than the cost of other water supply options considered. One lesson learned from the site was that algae growth in the source water may be problematic and lead to well clogging. Removal of the algae prior to ASR recharge would alleviate this problem.

Portland ASR site. The City of Portland Bureau of Water Works operates the largest municipal water supply in Oregon, serving approximately 840,000 people (Portland Bureau of Water Works, 2001). The primary source of water for the system is the Bull Run watershed located east of the city. The current system has an average day demand of 115 MGD with short-term peaks as high as 200 MGD. The City of Portland also relies upon the Columbia South Shore Well Field (CSSW) for backup water supply or supplemental summer supply when peak season demand requires it (Portland Bureau of Water Works, 2001). During the 1990s, the region grew steadily and its reliance upon the CSSW increased. A regional water supply plan included several ASR projects as components and the City of Portland decided to test the concept of ASR at its existing CSSW.

The ASR system is located at the CSSW and consists of four wells screened within the Portland area “Sand-and-Gravel Aquifer (SGA)”. The SGA is a highly permeable alluvial aquifer composed of medium to coarse sand and gravel. It has an average transmissivity of 2,700 ft²/day and storage coefficient of 2×10^{-4} as determined from aquifer performance tests. The aquifer is a confined aquifer in the study area. Four

additional wells screened within the Troutdale Sandstone are also under consideration. Water quality of the source water and ambient groundwater were deemed of acceptable quality. The source water from the Bull Run system is an unfiltered surface water supply, one of the few such systems remaining in the United States. The source water is disinfected with chlorine at the headworks, and later ammonia and sodium hydroxide are added to maintain a disinfectant residual and for corrosion control (Golder and Associates, 2003). TSS of the recharge water generally ranged from 1.0 to 1.67 mg/l, however, maintenance operations may have introduced highly turbid water into the well system in June 2002 (Golder and Associates, 2003). During recovery operations, minor amounts of disinfection by-products (total trihalomethanes) were observed but at concentrations well below regulatory levels. Minor geochemical enrichment of calcite and magnesium was also noted during ASR recovery operations resulting in a small increase in water hardness. In addition, during long-term testing of all four ASR wells, excessive sand generation was noted by the site operator. This was later determined to be due to lack of a pressure relief valve on the well or a faulty packer within the well. Dissolved oxygen levels were usually higher during initial recovery operations than during extended recovery. This may be due to biological growth within the wells. Recovery of injected water approached 100% and provided a high-quality water supply. Geotechnical problems resulting from ASR operation were explored at the site but found to be minimal. Subsidence of overlying clay confining units was not deemed a problem due to past geologic history and pre-consolidation from the weight of glacial ice. The author completed a thorough cost analysis of the project and determined that the overall unit cost was approximately \$0.29 per 1,000 gallons recovered. This unit cost was

significantly less than the cost of other water supply options considered by the City of Portland. One lesson learned from the site was that well clogging problems can materialize at any site if the source water TSS is high or if the TSS is low but the mass injected is high (Moncaster, 2004). Well clogging at the CSSW has resulted in a 5 to 37% reduction in well efficiency for the project. Periodic well re-development can help improve this situation but only if begun at the early-stages of a project.

Beaverton ASR site. The City of Beaverton ASR Project is located south of Portland, Oregon at the City's Sorrento Water Works (Groundwater Solutions, Inc., 2004). The project receives its water from the Joint Water Commission. The Joint Water Commission's treatment plant is located near Forest Grove, Oregon and receives its water from the Trask and Tualatin Rivers (Eaton, 2004). The ASR system consists of two wells that store water in the basalt aquifer during winter and spring months when river flows are high. The stored water is recovered in the summer and fall months to help meet peak system demands. The wells are also available during emergency situations such as extreme weather or flooding of the primary water supply. As of February 2004, six recharge and recovery cycles have been completed. During 2002 to 2003, 394.6 million gallons of treated drinking water were stored in the basalt aquifer by the ASR system.

The ASR wells are screened (open-hole interval) within the Columbia River Basalt Group (Eaton, 2004). The aquifer is highly fractured and behaves as a confined to semi-confined aquifer with average transmissivity of 13,369 ft²/day and a storage coefficient of 1×10^{-4} . Water quality of the source water and the ambient groundwater is deemed acceptable. The source water is chlorinated at the treatment plant and usually contains TSS of approximately 10 mg/l. During recovery operations, minor amounts of

disinfection by-products (total trihalomethanes) were observed but at concentrations well below regulatory levels. One interesting observation is that the concentration of total haloacetic acids (HAAs) was reduced to non-detectable levels during storage while the total trihalomethanes were reduced based upon mixing patterns of other compounds. It is possible that biodegradation of the HAAs did occur at the Beaverton ASR site. No significant geochemical reactions have been observed at the site, only simple mixing and hydrodynamic dispersion effects. Radon concentrations of the recovered water sometimes exceed the regulatory limit similar to the ambient groundwater supply. Aeration of the recovered water could remove this as an issue. No pathogens were observed in recovered water although minor occurrences of total coliform were noted in the source water. Dissolved oxygen levels were usually higher during initial recovery operations than during extended recovery. This may be due to biological growth within the wells caused via recharge of oxygen rich water. Recovery of injected water approached 100% and provided a high-quality water supply.

The City of Beaverton completed a thorough cost analysis of the project and determined that the overall unit cost was approximately \$1.99 per 1,000 gallons recovered. This unit cost was significantly less than the cost of other water supply options considered (Eaton, 2004).

There are many lessons learned from this ASR project site. First, during recharge events high water levels were noted in the aquifer. The high water levels were partly attributed to increased flow in an area seep. In an urbanized environment, seeps can be problematic due to perceived flooding risks. Monitoring of existing seeps before and during recharge events can aid in the interpretation of the seep flows. Second, the static

water level of the basalt aquifer is typically 200 feet below land surface, and without a down hole control valve, air entrainment can be problematic at the site. Third, lining old well boreholes can reduce turbulence during injection and reduce head loss during injection and recovery. Lastly, periodic back flushing of the wells is critical to maintaining efficient ASR well operations and reducing well clogging.

Salem ASR site. The City of Salem ASR project is located in Oregon in the southern Salem Heights area of the city (Golder Associates, 1995). Salem is in the northern end of the Willamette Valley between the Cascade and Coastal range of mountains. Salem is the state capital and home to approximately 140,000 residents. This population is expected to rise dramatically over the next decade and approach 228,000 by 2013 (Butts, 2002). Traditionally, most water supply in the area is provided through allocation of treated surface water from the Santiam River. This supply has been very reliable, however, severe flooding in 1996 caused the treatment sand filter plant to be shut down for several days (Butts, 2002). The flooding event and other circumstances have led city engineers to question the long-term viability of the existing water supply. Alternate water supply systems may be more dependable in emergency situations and could be located closer to demands than the current system. Due to these issues, the City of Salem is exploring ASR as a water supply option for emergency supplies and supplemental water for use during peak demand periods.

Pilot testing began in 1995 and included a significant data collection program. Pilot testing included long term pumping tests and a 30-day injection day where 38,315,700 gallons of treated water were injected into the ASR well. The ASR test well is drilled to 316 feet below land surface and includes a 12-inch diameter steel casing with an open-

hole interval of 36 feet. The ASR wells are screened (open-hole interval) within the Columbia River Basalt Group (Golder Associates, 1995). The aquifer is highly fractured and behaves as a confined to semi-confined aquifer with average transmissivity of 32,000 ft²/day and a storage coefficient of 2×10^{-3} . The aquifer is believed to have formed when molten lava flowed into a marshy depression or shallow lake. The contact of the lava and the water caused the lava to quench very rapidly causing extensive fracturing of the basalt rock (Golder Associates, 1996a). The rock surrounding the study area is postulated to have cooled over a long period of time, thus forming considerably less fractures in the basalt. This surrounding rock forms a natural barrier to groundwater flow and provides an excellent underground storage zone (Golder Associates, 1996a). Additionally, geologic faulting in the area may provide an additional barrier to groundwater flow.

Water quality of the source water and the ambient groundwater is deemed acceptable and is very similar (Golder Associates, 1996b). The source water is filtered through slow sand filters and disinfected with chlorine. It usually contains TSS concentrations of approximately 0.3 mg/l or less. The source water does contain minor amounts of total coliform and fecal coliform as well as low concentrations of disinfection by-products including THMs and HAAs (EPA, 1999b). Average concentrations of TTHMs and THAAs were 35 ug/l and 26 ug/l, respectively (Golder Associates, 1996c). During the 60-day storage operations prior to ASR recovery, minor amounts of a few disinfection by-products including chloroform and bromodichloromethane were observed but at concentrations well below regulatory levels. No significant geochemical reactions have been observed at the site, although extensive geochemical modeling did predict possible precipitation of ferrihydrite compounds (Golder Associates, 1996b), known to

cause well screen encrustation (Johnson, 1981). The modeling also concluded that only simple mixing and hydrodynamic dispersion effects would predominate the final chemistry of the mixed waters. Radon concentrations of the recovered water sometimes exceed the regulatory limit similar to the ambient groundwater supply. Aeration of the recovered water could remove this as an issue. No pathogens were observed in recovered water although minor occurrences of total coliform were noted in the source water. Bacterial “standard plate count” levels were usually higher during initial recovery operations than during extended recovery. This may be due to biological growth within the wells. Recovery of injected water approached 100% and provided a high-quality water supply.

Several interesting items were noted based upon a review of the Salem ASR reports. First, although the aquifer is quite transmissive and the injection water TSS is extremely low, minor well clogging was observed at the ASR well. Well efficiencies declined from 98% to 86% after 30 days of injection at 1.3 MGD. Clogging was attributed to the TSS and possible air entrainment of the aquifer. Second, geotechnical evaluations were completed at the site to evaluate the possibility of vertical hydraulic “jacking” or hydrofracturing. It was recommended that the allowable head increase be limited to 200 feet of buildup to minimize jacking effects (Golder Associates, 1996b).

Salt Lake City ASR site. The southeast Salt Lake County ASR project is one of 13 demonstration projects implemented by the Bureau of Reclamation (BOR) under the “High Plains States Groundwater Demonstration Program Act of 1983”. The project was cost-shared between the BOR and the Salt Lake County Water Conservancy District (SLCWCD) starting in 1990. EPA and USGS provided technical assistance and

interpretation of the data. The project is located in a highly urbanized residential/commercial area in Salt Lake County, about 8 miles SE of Salt Lake City, Utah (Bureau of Reclamation, 1996b).

The project was developed to test the feasibility of providing needed additional capacity for delivery of municipal water by storing treated drinking water underground during the winter for withdrawal during the peak summer demand season. The project, if found feasible, would be expanded in lieu of enlarging water conveyance facilities or building additional costly surface storage. Injection wells were selected for use since spreading basins were not practical in the urbanized area. Source water was available from Bureau of Reclamation's Deer Creek Reservoir in Provo Canyon, UT via the Provo River and then the 34-mile long Salt Lake Aqueduct. The water is then conveyed to the Little Cottonwood Water Treatment Plant.

The geological environment is an unconfined aquifer composed of cobbles, sands, and silts deposited as colluvium. The site location takes advantage of high transmissivity values determined to be approximately 4,947 ft²/day at the project site. Natural recharge to the aquifer is derived from Wasatch Front precipitation (primarily snowmelt) percolating through fractured bedrock into the unconfined aquifer nearby. The groundwater quality is excellent.

The project consists of one injection well, one recovery well, one dual-purpose well, three monitoring wells, and an inline filtration plant with dual-media pressure filters and UV disinfection. Other existing wells may also be utilized for recovery and monitoring. Recovered water is distributed through existing mains owned by SLCWCD. The injected water meets all required regulatory standards under the UIC program. The

source water is chlorinated at the main treatment plant and usually contains TSS of approximately 1.0 mg/l or less. During recovery operations, minor amounts of disinfection by-products (total trihalomethanes) were observed but at concentrations well below regulatory levels.

Well plugging problems were mainly attributed to small suspended particles that passed through the dual media filters. In addition, residues from water treatment coagulation chemicals may precipitate in the aquifer. Plugging at one injection well created an additional 30 feet of head loss. Calculations suggest that up to 10% of the specific capacity is lost over the injection period. Redevelopment activities included mound collapse, wire brushing the well casings, swab surging, and pump surging. Pump surging was found to be the most effective method. One-hour redevelopment cycles were required to return most of the injection well capacity. Redevelopment resulted in substantial unanticipated costs.

The Bureau of Reclamation and City of Salt Lake completed a thorough cost analysis of the project and determined that the overall unit cost was approximately \$0.09 per 1,000 gallons recovered. This unit cost was significantly less than the cost of other water supply options considered (Bureau of Reclamation, 1996b).

One important lesson learned at this site was that during the project, SLCWCD met with greater than expected legal and institutional demands due to debates concerning water rights. One conclusion was that the State Water Engineer did not have existing state laws to address artificial groundwater recharge projects. Regulatory agencies did not know how to address the project causing the use of significant staff time from the

sponsor and regulators. In addition, local zoning laws made the acquisition of project sites time consuming.

Huron AR site. The Huron artificial recharge and recovery project is one of 13 demonstration projects implemented by the Bureau of Reclamation (BOR) under the “High Plains States Groundwater Demonstration Program Act of 1983”. The project was cost-shared between the BOR and the South Dakota State University starting in 1990 (Bureau of Reclamation, 1996d).

Multiple droughts have been recorded within the James River Basin in eastern South Dakota over the last 20 years. These droughts forced local irrigation users to utilize the local glacial aquifers heavily causing persistent water declines within the aquifer system. In addition to the existing irrigation wells in the study area, the City of Huron, South Dakota owns a municipal well field in the area that is used for backup water supply when low flows are present within the James River. The well field is mostly used during late winter if the James River is frozen or in late summer or early fall when river flows are low. The City of Huron anticipates continuing demand for the limited water supplies available. This problem was seen as an opportunity to evaluate artificial recharge in this area to test the overall recharge potential for the buried glacial sediments (Bureau of Reclamation, 1996d).

The concept plan developed was to use high flows from the James River during high spring flow events, treat the source water in the City of Huron’s water treatment plant, pipe the water to the well field, and then inject the water into the Warren aquifer, a buried glacial aquifer in the study area. A portion of the injected water was later recovered for water supply purposes (Bureau of Reclamation, 1996d).

The glacial deposits of the Warren aquifer consist of both till and outwash. Till is a heterogeneous mixture of silt, sand, gravel, and boulders in a clay matrix. Outwash consists of cross-bedded sand, gravel, and silt. In the project area, the outwash has a thickness from 10 to 75 feet. The confined Warren aquifer transmissivity based upon literature values provided by Domenico and Schwartz (1998) is estimated to be 3,000 to 5,000 ft²/day.

The source water from the James River is generally good quality water but occasionally suffers from agricultural runoff that has caused nitrates and arsenic to be detected in river water samples (EPA, 1999b). In addition, fluoride concentrations routinely exceed regulatory standards. The ambient groundwater quality is fair to poor with high concentrations of sodium, sulfate, iron, and manganese. Recovery of injected water was not accomplished during the test project so data on disinfection by-products or possible geochemical issues are unknown.

Well clogging of the injection wells was observed at the project site. Post-recharge aquifer testing revealed a transmissivity decrease of approximately 20%. This decrease was attributed to air entrainment during injection and to minor physical clogging in the well filter packs (Bureau of Reclamation, 1996d).

An economic analysis completed by South Dakota State University estimated long-term costs of the project to approximately \$0.29 per 1,000 gallons injected. The unit costs of recovered water were not included in the estimate since recovery operations have not yet commenced. The total unit cost to recharge and recover the water is estimated to range from \$0.35 to \$0.50 per 1,000 gallons.

Several lessons learned are available from this test project. First, naturally high water tables or abnormally high precipitation can limit recharge operations. Second, artificial recharge can improve the water quality of the ambient groundwater, ultimately resulting in reduced water treatment costs at the well field.

Washoe County ASR site. The Washoe County, Nevada artificial recharge and recovery project is one of 13 demonstration projects implemented by the Bureau of Reclamation (BOR) under the “High Plains States Groundwater Demonstration Program Act of 1983”. The project was cost-shared between the BOR and Washoe County, Nevada starting in 1990 (Bureau of Reclamation, 1996a). The project is located in the northwestern Reno/Sparks, Nevada urban area.

Washoe County, Nevada is located in a semi-arid area in the shadow of the Sierra Nevada Mountains. Meeting water supply needs in the Reno/Sparks metro-area has become increasingly difficult. The primary source of drinking water is from the Truckee River, which has been characterized by wide flow variations depending upon precipitation in the mountains. When flows are high, a surplus of water supply exists; when flows are low, groundwater is used to supplement water supply needs. This continual groundwater use has led to persistent groundwater level declines within the study area. Artificial recharge of excess storm water would provide a means to restore natural groundwater levels in the area. In addition, in the future, recovery of the recharged water would also be possible.

The project is relatively simple and includes a booster pump station, several pipelines, carbon bed filters, and injection wells. Excess Truckee River water is diverted approximately 10 miles from Golden Valley and 15 miles from Lemmon Valley

(suburban communities in Reno, NV) and brought via pipeline to the two communities for injection into the aquifer.

Limited hydrogeologic information was available in references reviewed for this project. The aquifer system consists of glacial sands and gravels in cross-bedded units. Aquifer transmissivities are estimated to range from 1,000 to 5,000 ft²/day. Related to the overall site geology, geotechnical issues were evident during the 1995 recharge cycle. The winter of 1994/1995 was very wet with snow pack approaching 200% above average (Bureau of Reclamation, 1996a). During this recharge cycle, injection pressures were gradually increased to improve injection rates and overall system throughput. In the Lemmon Valley portion of the project, pressures were increased to 43 psi. At this pressure, water began percolating to the ground surface along distinct radial soil fractures around a nearby well. The soil crack was attributed to subsidence from long-term pumping combined with aquifer dewatering. Due to the problem, pressures were reduced to a few psi, severely limiting recharge operations.

The water quality of the source water within the Truckee River is excellent with low total dissolved solids, metals, and color. The water is treated with sand filters and chlorination before being piped to the injection wells. At the injection wells, the water flows through carbon filters prior to injection to minimize TSS and total organics in the recharge water (Bureau of Reclamation, 1996a). This process minimizes the generation of disinfection by-products, although TTHMs were observed at concentrations of 60 mg/l at the injection well and 28 mg/l in one of the well field production wells. Concentrations of HAAs were minimal. The TSS is generally less than 2.0 mg/l during recharge operations. The ambient groundwater contains several compounds that exceed

secondary maximum contaminant levels for taste and odor including sulfate, iron, manganese, zinc, copper, color, and total dissolved solids. In addition, minor amounts of naturally occurring radionuclides were detected in the ambient groundwater. No geochemical issues have been reported at the site.

An economic analysis completed by Washoe County estimated long-term costs of the project to be approximately \$1.13 per 1,000 gallons injected. The unit costs of recovered water were not included in the estimate since recovery operations have not yet commenced (Bureau of Reclamation, 1996a). The total unit cost to recharge and recover the water is estimated to range from \$1.30 to \$1.50 per 1,000 gallons.

The most important lesson learned in this project is that competing demands for water exclude the concept of available “excess” water for recharge. Even during high flow events, water that might be considered as excess has a demand. Environmental flows to support the endangered Cui-ui sucker native to Pyramid Lake (which lies as the main discharge point for the Truckee River) compete with other water interests representing urban areas, irrigation interests, and the Paiute Tribe of Indians. Water rights are scarce and are the subject of intense litigation. Therefore, in the case of the project, the most important project consideration is institutional constraints stemming from over appropriation of the Truckee River (Bureau of Reclamation, 1996a).

Last Vegas ASR site. The Las Vegas ASR system is currently one of the largest in the world with more than 40 dual-purpose ASR wells (Landmeyer et al., 2000), each capable of recharging one to three MGD. The Las Vegas Valley is in southern Nevada, about 20 miles west of Lake Mead and the Colorado River (Katzner and Brothers, 1989). Much of the area is extremely arid receiving only 4 inches of precipitation annually. The

surrounding mountain ranges may get up 20 inches of precipitation in the form of mostly snow cover. The City of Las Vegas has grown rapidly in recent decades and in 1987, the City was facing water supply problems. Most of the water supply comes from treated Colorado River water (70 % to 80 %) and ground water pumping. As a result of over drafting the groundwater aquifer in the area, the groundwater source was becoming depleted. Therefore, water managers sought to bring Colorado River water into the valley during the low-demand winter months and store the water underground using an ASR system (Katzner and Brothers, 1989).

The ASR project study area is located on top of extensive basin fill sediments. Coarse-grained sediments including sand, gravel and pebbles are intermingled with silt and clay. The depth of these sediments is approximately 1,000 feet. The aquifer units within this massive fill zone are characterized as unconfined to semi-confined with highly variable transmissivities ranging from 1,000 to 40,000 ft²/day (Pyne, 1995).

Most of the groundwater recharge emanates from the surrounding mountains such that the groundwater quality is characterized as carbonate water with a very high hardness (Katzner and Brothers, 1989; Pyne, 1995). The groundwater has considerable amounts of bicarbonate, calcium, magnesium, and sulfate. The treated Colorado River source water is characterized as reasonably good with high concentrations of sodium, sulfate, and nitrates. The treated water is disinfected via chlorination creating moderate amounts of disinfection by-products including 50 ug/l of TTHMs and 15 ug/l of HAAs, on average (Landmeyer et al. 2000). In addition, the treated source water contains up to 1 mg/l of free chlorine residual and 3 mg/l total organic carbon (TOC), leading to in-situ formation of disinfection by-products. Upon initial recovery of the stored surface water,

TTHM and THAA concentrations rose to 85 ug/l and 20 ug/l, respectively (Landmeyer et al., 2000). According to Miller et al. (1993), dissolved organic carbon (DOC) serves as the actual precursor material for by-product formation and most treatment plants do not remove the entire DOC. Unfortunately, sorption of by-products onto aquifer materials at the site was found to not occur (Miller et al., 1993). Also, certain trihalomethanes such as chloroform were found to be quite resistant to biodegradation, however, biodegradation of HAAs was found to be substantial (Landmeyer et al. 2000). Geochemical studies at the site initially indicated that calcite precipitation would be problematic, however, during pilot testing, this was not found to be a problem at all (Pyne, 1995).

An economic analysis completed by the author estimated long-term unit costs of the project to be approximately \$0.25 per \$0.75 per 1,000 gallons recovered. The costs were estimated based upon standard water treatment costs, well costs, and pipeline costs at similar projects.

The most important lesson learned at this project is that management of the chlorine residual during disinfection of ASR source water is critical. If the residual is too high, in-situ formation of disinfection by-products may occur. In addition, another important conclusion from the project is that biodegradation of HAAs does occur during ASR storage, reducing this as a problematic issue. Unfortunately, THMs such as chloroform were found to be resistant to biodegradation.

Calleguas ASR site. The Calleguas ASR project is administered by the Calleguas Municipal Water District, a member agency of the Metropolitan Water District of Southern California (Calleguas, 2004). Calleguas serves 550,000 water users in southern

Ventura County, California including the cities of Oxnard, Camarillo, Thousand Oaks, and Simi Valley. All of the potable water supply provided by Calleguas is derived from surface water imported from the California State Water Project. This water is treated at a large Metropolitan treatment in Granada Hills. Treated water is then conveyed through a series of pipelines and tunnels to water users (Calleguas, 2004). The current project includes 26 ASR wells capable of approximately 64 MGD of total capacity. Each well is located 800 to 1,200 feet deep within the Fox Canyon Aquifer. All wells are equipped with vertical turbine pumps with motors ranging from 600 to 800 horsepower. The entire system can be remotely operated through a complex instrumentation system (Calleguas, 2004). Wolcott (2003) reports that the system is also designed to provide emergency water supply for the region in case of an earthquake.

The ASR wells are constructed in a hydrogeologic basin filled with alternating layers of marine sands, marine gravels, and marine silts/clays (Pyne, 1995). The Fox Canyon Aquifer is the primary storage zone for the ASR wells and consists of a confined aquifer composed of 200 to 400 feet of marine sand and non-marine sand and gravel. The aquifer is highly productive with aquifer transmissivities ranging from 600 to 16,000 ft²/day. Pyne (1995) reports an average value from initial site testing of 19,385 ft²/day along with a storage coefficient of 4×10^{-6} . Pyne (1995) also reports an aquifer porosity of 23% and a dispersivity of 22 feet. During cycle testing, the test well specific capacity was noted to decline due to well clogging and the low temperature of injection water. The 11 degree Celsius temperature differential between the recharge water and the ambient groundwater resulted in an apparent transmissivity lower than initial testing suggested.

The recharge water quality is characterized by moderately high TOC of 2.9 mg/l, high concentrations of nitrate, chloride, and boron. The recharge water also contained up to 74 ug/l of TTHMs (Pyne, 1995) and 3 mg/l of TSS. The ambient groundwater contains high concentrations of potassium, manganese, iron, hardness, and radon. During storage of the cycle test water, TTHM concentrations were reduced and iron and manganese concentrations dropped also possibly indicating precipitation or some other geochemical reaction. Unfortunately, during recovery, iron and manganese concentrations still remained higher than desired by Calleguas. Therefore, Calleguas is considering additional treatment of the constituents (Calleguas, 2004). Lastly, moderate concentrations of radon were observed in some recovered water data.

An economic analysis completed by the author estimated long-term unit costs of the project to be approximately \$0.51 to \$0.65 per 1,000 gallons recovered. The costs were estimated based upon standard water treatment costs, well costs, and pipeline costs at similar projects. Wolcott (2003) reports a value of \$0.72 per 1,000 gallons recovered compared to a nearby dam project where the costs are 10 to 15 times higher per gallon. One interesting economic item to note is that each vertical turbine pump is designed to produce electricity as water is injected. According to Wolcott (2003), the amount of power produced is more than enough to run the project and excess power is sold back to the power grid. This is an important benefit of the project.

The most important lesson learned at this project is that of well clogging management. Well clogging can be severe since the various supply pipelines and tunnels can get entrained with rust and sand. These materials can lead to major well clogging during recharge operations (Pyne, 1995). A regular program of well re-development is

usually required at the site to maintain high inflow rates. In addition, site operations revealed that flushing the well and surrounding pipelines to waste for a short duration before recharge or recovery operations can reduce well clogging problems.

Lancaster ASR site. The Antelope Valley ASR project is located near Lancaster, California. In order to restore declining water levels in the aquifer in the study area, an ASR demonstration project began in 1995 (Baqai, 2002). Surface water from the California State Water Project was imported and treated prior to injection into the ASR system. Antelope Valley is located at the western end of the Mojave Desert (Fram et al., 2003) where precipitation is very low and aquifer recharge is minimal. Therefore, artificial recharge can be very effective.

The Lancaster sub-basin contains alluvial and lacustrine deposits that can be as much as 5,000 feet thick (Fram et al. 2003). The alluvial deposits consist of interbedded zones of silt, sand, and gravel (Leighton and Phillips, 2003). At the ASR site, the upper unconfined aquifer extends from the water table to a depth of 510 feet below land surface. The semi-confined middle aquifer extends from 510 to 730 feet below land surface, and the confined deep aquifer extends from 870 feet to the surface of bedrock (Fram et al, 2003). The ASR wells penetrate the upper and middle aquifers that have an average aquifer transmissivity of approximately 2,500 ft²/day (Leighton and Phillips, 2003).

The water quality of the surface water is characterized by moderately high concentrations of chloride, nitrate, boron, and TOC. The treated source water also contains significant quantities of TTHMs of up to 75 ug/l (Fram et al. 2003) and residual free chlorine with a mean concentration of 0.79 mg/l. The ambient groundwater quality

is deemed acceptable. THMs including chloroform, CHCl_3 , bromodichloromethane, CHCl_2Br , dibromochloromethane, CHClBr_2 , and bromoform, are formed through reactions between natural dissolved organic carbon present in water and chlorine. THMs are carcinogenic compounds. Previous testing from past cycles has revealed that recovered water still contained measurable concentrations of THMs long after continuous pumping had ended (Fram et al. 2003).

Lab studies of biodegradation indicate that aquifer bacteria did not degrade CHCl_3 or CHBr_3 under aerobic conditions but did degrade CHBr_3 under anaerobic conditions. Unfortunately, the aquifer is aerobic in nature and the dominant form of THM is CHCl_3 so minimal biodegradation can be expected. THM formation in the aquifer following injection is controlled by the amount of residual chlorine in the injectate.

Lancaster site data clearly show a decreasing trend of peak THM values over 3 cycles of recharge and recovery. All THM data in monitoring wells have indicated max concentrations of 75 ppb of THM that have declined over time. Total residual chlorine in the recharge water ranged from 0.7 to 1.38 mg/L, while free residual ranged from 0.5 to 1 mg/l. Fram et al. uses the molar ratio of bromine to chlorine to evaluate reaction kinetics in the aquifer. Fram et al. discuss THM formation and offer the following mechanisms that control formation potential:

- Reaction-limiting concentration of chlorine
- Reaction-limiting concentration of DOC
- The propensity of DOC to form THMs
- Contact time between the DOC and chlorine
- The ratio of concentrations of bromide and DOC
- PH

Fram et al. also report that bacterial density in and around the ASR wells apparently increased following injection of ASR recharge water. This may be due to

DOC in water that can be utilized as a food source for the bacteria. It is surmised that further acclimation of the onsite bacterial community could result in CHBr_3 biodegradation in future ASR cycles. Well clogging was not discussed in the references reviewed, however, it is expected to be an important operational consideration at the site. Economic data also were not available in the literature reviewed for this report.

The most important lesson learned at the site is that management of the chlorine residual during disinfection of ASR source water is critical. If the residual is too high, in-situ formation of disinfection by-products may occur. In addition, some THMs can be biodegraded during ASR storage including bromoform (under anaerobic conditions), however, chloroform appears quite resistant to biodegradation.

Denver ASR sites. Two notable ASR projects are located in the Denver, Colorado area. The Centennial Water and Sanitation District administer the ASR system, located at Highlands Ranch. Centennial provides water to 22,000 customers in the exclusive Highlands Ranch planned development located in the south Denver area (Pyne, 1995). The ASR system is designed to aid Centennial in meeting its peak summer water demands that are 217% above average annual water demand (Pyne, 1995). The ASR system is completed within the Arapahoe formation. The ASR storage zone is located at a depth of 922 feet below land surface.

The Arapahoe aquifer is a confined artesian aquifer consisting of loosely cemented sands and sandstone. The aquifer has an average transmissivity determined from aquifer tests of $1,000 \text{ ft}^2/\text{day}$ and a storage coefficient of 3×10^{-4} . The Highlands Ranch site is notable in that it probably has the lowest transmissivity of operational ASR sites as of 1993 (Pyne, 1995).

Water quality of the source water and the ambient groundwater are fair. The quality of both source and ambient water is very similar with the exception of iron and manganese which are both more concentrated in the groundwater. Geochemical investigations indicated a plugging potential due to calcium carbonate and ferric hydroxide precipitation (Pyne, 1995). The actual testing of the ASR wells did not indicate any geochemical problems. The testing results also revealed substantial reductions of TTHMs and TOC during ASR storage. Data on total suspended solids was not available but is assumed to be less than 2 mg/l based upon the water treatment system utilized prior to injection. According to data presented by the EPA (1999b), the source water also had a high combined radium 226 and 228 concentration since a portion of the source water is derived from groundwater sources.

Due to well clogging and the low aquifer transmissivity, water levels rose almost 410 feet during cycle 3 recharge operations resulting in likely hydraulic fracturing of the aquifer. This conclusion presented in Pyne (1995), was validated further during the following recharge and recovery cycle where hydraulic response due to the aquifer was much less pronounced indicating that fracturing had increased local-scale hydraulic conductivity. Economic evaluations of this system were not available in the references reviewed for this report.

Lessons learned from this site are many and varied. First, air entrainment due to cascading recharge water was very problematic. This problem was reduced once a down hole control valve was utilized in the ASR well. Another lesson learned is that well clogging combined with low permeability storage zones can lead to geotechnical problems including hydraulic fracturing of an aquifer. Lastly, the exact causes of well

clogging can vary so proper data collection during ASR testing is essential to determining the root causes of any observed well clogging.

The second Denver ASR project is one of 13 demonstration projects implemented by the Bureau of Reclamation (BOR) under the “High Plains States Groundwater Demonstration Program Act of 1983”. The project was cost-shared between the BOR and a team from the Willows Water District and Denver Water starting in 1990 (Bureau of Reclamation, 1997).

The ASR project is located in the southeastern section of the greater Denver Metropolitan area along the mountains of the Colorado Rockies. The area is semi-arid with an average precipitation of 15 inches per year. The project was envisioned as a way to store excess treated surface water during periods of surplus and recover the stored water during periods of peak demand (Bureau of Reclamation, 1997). The source surface water is derived from the South Platte River where flow varies widely from 75,000 acre-feet to 800,000 acre-feet (Bureau of Reclamation, 1997).

The ASR well system is completed in the permeable portions of the confined Arapahoe aquifer. The Arapahoe aquifer generally consists of a 400 to 700 feet thick sequence of interbedded sandstones, siltstones, and shale. Water levels have declined over time due to over allocation of pumping (Bureau of Reclamation, 1997). The average transmissivity of the aquifer at the project site is 856 ft²/day (Aikin and Turner, 1987).

The quality of the source water and ambient groundwater is excellent. Potential water quality concerns investigated during the ASR testing program included geochemical precipitation, clay dispersion, and geochemical redox reactions. In addition, due to the introduction of highly oxygenated surface water into the aquifer, microbial

growth was also a major concern. During operations that included 34 separate recharge and recovery cycles, no adverse geochemical problems were recorded (Bureau of Reclamation, 1997). Minor well clogging did occur but could be relieved through periodic back flushing. Initial cycle testing indicated that well re-development was most efficient when at least 0.5 to 1% of the quantity of recharged water is recovered (Aikin and Turner, 1987). The project also revealed that recovery efficiency of the project approached 98%.

An economic analysis of the project completed by the local sponsor and the Bureau of Reclamation determined that the unit cost to recharge and recover the stored water was approximately \$2.45 per thousand gallons. Approximately \$1.65 of this amount was the cost to purchase the imported surface water for recharge. The conclusion of the economic analysis was that the project was feasible if surface water supply is available at a reasonable unit cost and is located reasonably close to the project site (Bureau of Reclamation, 1997).

There are several lessons learned from the project. First, the need to control air entrainment of the recharge water through back pressure or a down hole control valve is very important to minimize one cause of well clogging. Second, the potential negative hydraulic impact of recharging source water significantly colder than the ambient groundwater should be explored prior to the start of ASR testing. Colder recharge water will lower the apparent aquifer transmissivity leading to reduced hydraulic efficiency. Institutional issues are an important consideration in an area where the legal framework does not include means to adjudicate artificial recharge projects.

Alamogordo ASR site. The City of Alamogordo ASR project is located in Alamogordo, New Mexico at the existing La Luz well field (Livingston and Finch, 2003). The project includes recharge of excess spring water into a modified well field well. The well recharge is accomplished using gravity flow into the aquifer. During initial testing 21 million gallons of treated spring water were injected into the ASR storage zone. The City of Alamogordo has existing water right to the spring water and uses 76% of its allocation annually. The ASR program would seek to recharge the remaining 24% of the spring flow allocation (Livingston and Finch, 2003).

The aquifer is a basin fill aquifer within the Tularosa Basin. It is hydrogeologically closed due to the geometry of the surrounding mountains making it an excellent candidate for ASR storage. The basin sediments may be as thick as 3,000 feet in the study area underlain by limestone bedrock. Much of the area recharge emanates in the nearby mountains and then discharges into the basin fill sands, gravel and boulders. The average transmissivity in the study area was estimated through groundwater modeling to be approximately 2,500 ft²/day.

The water quality of the source spring water is outstanding with respect to TSS, metals and TOC. The groundwater quality is poor with TDS values of up to 2,500 mg/l and objectionable concentrations of hardness (Livingston and Finch, 2003). Water quality evaluations reviewed the potential for clay dispersion in the aquifer as well as geochemical problems with no issues being raised that would limit project feasibility. Minor precipitation of iron or manganese on the ASR well screen was identified as a problem and did occur during ASR pilot testing (Livingston and Finch, 2003). Well clogging was not a major problem and air entrainment was minimized through the use of

a drop pipe or eductor that ensured positive pressure during recharge operations. Long-term predictions of water recovery efficiency were estimated to be 85%.

An economic evaluation of the project was completed for the project sponsor. It was determined that the project would be feasible and low cost with an average unit cost of \$0.10 per thousand gallons recovered.

A valuable lesson learned from this project is that ASR projects may allow users to fully utilize available water allocations. In addition, institutional water rights issues need evaluation for any project. Water rights questions noted by Livingston and Finch (2003) include:

- Can diverted surface water legally be stored in an aquifer ?
- Can the stored surface water be recovered based on the existing water right ?
- How is the stored water protected from other potential appropriators ?

Green Bay ASR site. The Green Bay ASR project is located in Green Bay, Wisconsin within an emergency water supply well field utilized by the Green Bay Water Utility (CH₂M Hill, 1999). The Green Bay Water Utility (GBWU) supplies water to industry and to more than 96,000 people in the Green Bay area. Historically, GBWU used water supply wells to provide this water but serious water level declines in the aquifer led the utility to abandon the existing well field in favor of surface water from Lake Michigan (CH₂M Hill, 1999). The well field is now only used for emergency supply purposes. The daily demand of water in the area is expected to rise from 18.2 MGD to 25 MGD by the year 2030. In addition, the peak to average daily demand is expected to rise from 1.48 to 1.84 by 2030 (CH₂M Hill, 1999). In order to respond to growing water needs, the GBWU has been investigating the use of ASR to provide additional water supply capability.

The hydrogeology in the study area is quite complex. Ancient bedrock formations of sandstone underlie the area and are hydraulically connected, forming one large confined aquifer (CH₂M Hill, 1999). The Sandstone aquifer is confined between shale on the top and granite bedrock below. Individual water-bearing units have been identified including the Elk Mound Sandstone, the Prairie du Chien Group, and the St. Peter Sandstone (CH₂M Hill, 1999). Aquifer performance testing completed in 2000 provided transmissivity data for the entire aquifer system as well as each unit (CH₂M Hill, 2000c). Constant rate pump testing of the entire mix of units indicates a transmissivity of 655 to 1,200 ft²/day. Aquifer pump testing of just the Elk Mound/St. Lawrence provides an estimated transmissivity ranging from 600 to 1,600 ft²/day with an average storage coefficient of 5.5×10^{-4} . Similar testing for the St. Peter aquifer showed the transmissivity ranging from 388 to 1524 ft²/day with similar storage coefficient values (CH₂M Hill, 2000c).

The treated Lake Michigan water is generally of good quality. It is characterized by low average TDS of 154 mg/l, a slightly alkaline pH of 8.0, and is highly oxidic with concentrations of dissolved oxygen of 10 to 12 mg/l. The source water also contains minor amounts of nitrate, TOC (1.6 to 2.0 mg/l), TTHMs, HAAs, and radionuclides (CH₂M Hill, 1999). TSS is not measured regularly but is usually less than 1.0 mg/l. The ambient groundwater on the other hand, contains high concentrations of TDS, fluoride, iron, manganese, arsenic, and radionuclides including radon (CH₂M Hill, 1999). Additional water quality testing of the aquifer revealed similar results. Aquifer testing of production well # 10 revealed high concentrations of nickel (270 ug/l), radon (483 pCi/l), and arsenic (67 ug/l) (CH₂M Hill, 2000b). Both the radon and arsenic levels significantly

exceeded approved regulatory levels. Since high levels of heavy metals are naturally present in the groundwater, detailed geochemical testing, sampling, and modeling was completed.

The main geochemical problem is the presence of pyrite minerals within the sandstone matrix. Mineralogical studies identified multiple metals in the natural rock. The mineralogical studies included the collection of rock cores submitted for mineralogic testing to assess the amount and type of clay, the amount and form of pyrite and other minerals, the effective porosity, and the vertical and horizontal permeability (CH₂M Hill, 2000a). Testing included the following:

- X-ray diffraction mineralogical analysis (XRD)
- Thin Section Petrography
- Scanning Electron Microscopy (SEM)
- Acid insoluble residue analysis
- Cation exchange capacity and leachate analysis
- Laser or sieve particle analysis
- Helium porosity, air permeability, and grain size density
- Air permeability analysis (vertical plug orientation)
- Energy dispersive x-ray analysis
- Specific gravity analysis
- Core slabbing macroscopic core description and photography

Sections chosen for arsenic analyses were generally those that contained visible metallic deposits as well as sections that had a higher porosity that would be representative of the areas in each aquifer where the majority of water would flow. Observed arsenic concentrations in Wisconsin have previously revealed that the Elk Mound geologic unit

(sandstone with silt seams) had the highest concentrations of arsenic at 7.86 mg/kg. Prairie du Chien dolomite had 2.93 mg/kg of arsenic in rock mass. The worldwide average for arsenic in limestone is 2.6 mg/kg and sandstone 4.1 mg/kg (CH₂M Hill, 2000a).

ASR cycle testing at the site started in June 2002 and included 10 million gallons of recharge at production well # 10 (CH₂M Hill, 2003a). Following 22 weeks of storage, recovery operations commenced on December 2, 2002. Although at least three THMs appear to have degraded during storage, bromodichloromethane was contained within the recovered water. In addition, high concentrations of arsenic, nickel, and cobalt were also present in the recovered water. More than 1,200 percent of the recharged water had to be recovered from the well before arsenic concentrations fell below 10 ug/l (regulatory level) (CH₂M Hill, 2003a). Geochemical reactions between the pyrite and the recharged water seem to be very problematic at the site. A major reason for problems could be the supersaturated dissolved oxygen levels recharged into the aquifer. The source water naturally has high oxygen values but the oxygen level is further increased due to the disinfection method that is ozonation. This produces water with 15.5 mg/l of dissolved oxygen.

An economic analysis of the project has not been completed but it was noted that the cost of the utility to provide water using an ASR system was only 3.1 million dollars versus almost 39 million dollars for other alternatives (CH₂M Hill, 1999). However, due to persistent water quality issues in the recovered water, the project was discontinued. Additional studies of the geochemistry are planned to determine if any ASR project could be feasible.

A major lesson learned at this project site is to complete a thorough geochemical investigation in areas known to have existing problems. Water quality data from aquifer pump tests should have forewarned the site owner that the well # 10 site was going to be problematic. Another lesson learned is that at sites with potential pyrite problems or similar oxidation-reduction issues, ozonation should not be utilized as the primary disinfection process. Ozonation introduces additional dissolved oxygen that could exacerbate geochemical reactions in the aquifer.

Oak Creek ASR site. The Oak Creek ASR project is located just south of Milwaukee, Wisconsin in the town of Oak Creek. Oak Creek is a bedroom community of Milwaukee and is expected to experience considerable population growth over the next 20 years. The Oak Creek community was serviced by groundwater supply wells until 1976 when a large water treatment plant came online to provide treated water from Lake Michigan (Miller, 2001). The Oak Creek Water and Sewer Utility (OCWSU) administers the ASR project and brought the project to fruition through a thorough demonstration project completed in partnership with the American Waterworks Association Research Foundation (AWWARF). The demonstration test program was conducted to assess the feasibility of using ASR as a cost-effective alternative to other facility expansion options being considered by OCWSU (Miller, 2001).

The ASR project utilizes several of the existing historic water supply wells that are now only used in emergencies. The ASR project would aid with addressing critical peak water demands in the study area that are expected to grow. The ratio of maximum day demand to average day demand was approximately 1.32 in 1982. By 1994 it had increased to 1.98, and it is expected to rise again in the future (Miller, 2001). The

planned ASR scheme would ultimately allow the 20-MGD surface water treatment plant to provide the equivalent capacity of 28 MGD. This equates to an ASR design storage volume required of 240 MG in place by the year 2020.

Three main aquifers are present in the Milwaukee, Wisconsin area including Pleistocene age sand and gravels and glacial drift, the Silurian age Niagara dolomite, and sandstones of Cambrian and Ordovician age (Foley et al. 1953). Existing well logs at the Oak Creek project depict 150 feet of mostly, fine-grained unconsolidated Pleistocene deposits that do not form a major water-bearing unit (Miller, 2001). The Niagara dolomite is a massive, white to gray dolomite approximately 220 feet thick that transmits groundwater through secondary permeability features. It is a mostly unconfined aquifer in direct connection to the overlying Pleistocene deposits (Foley et al. 1953). The sandstone aquifer is the most significant aquifer in the study area. It is a confined aquifer that is confined by the Maquoketa shale formation on top and low permeability bedrock units at the bottom (Foley et al. 1953). Beneath the Oak Creek study area, the permeable aquifer begins 880 feet below ground with the St. Peter sandstone, which is approximately 110 feet thick (Miller, 2001). The transmissivity of the sandstone aquifer is variable so an onsite aquifer performance test was completed at ASR demonstration well # 1. Later aquifer testing was completed on well # 3. In addition, aquifer transmissivities were calculated after each of the four demonstration cycle tests were completed. The average transmissivity determined from these tests was 3,300 ft²/day with a storage coefficient of 2×10^{-4} . Porosity was estimated to range from 15 to 30% (Miller, 2001).

The water quality of the source water is considered excellent with low TSS (average of 0.021 mg/l), moderate dissolved oxygen (5.43 to 13.1 mg/l), moderate TTHMs (15 to 20 mg/l), moderate chlorine residual (1 to 1.4 mg/l), and low TDS (140 to 220 mg/l).

The groundwater quality is fair with objectionable amounts of iron, manganese, TDS, and radionuclides (mostly combined radium) (Miller, 2001). Generally, the temperature of the recharge water was 5 to 7 degree Celsius colder than the temperature of the groundwater. During recovery mixing of the recharged water and the ambient groundwater was measured using various conservative ions. Monitoring of TTHMs revealed that after longer storage times in the aquifer, the various THM components are degraded anaerobically with dibromochloromethane degrading rapidly and chloroform and bromodichloromethane degrading more slowly. In addition, geochemical oxidation of pyrite has been identified as another important issue at the site (Miller, 2001). Oxidation of pyrite was revealed through the increasing concentration of manganese in the recovered water. The ASR recovery efficiency of the project was nearly 100% by test cycle # 4. Moderate well clogging was experienced during cycle 1 but was reversed through daily back flushing cycles.

An economic analysis completed by OCWSU estimated long-term costs of the project to be approximately \$0.11 per 1,000 gallons recovered. The unit costs of recovered water do not include any required future treatment of manganese although this is not expected to be a major problem.

There are a few important lessons learned at this site. First, ASR can be an ideal alternative for meeting peak water demands and much cheaper than other options. Second, the geochemical reactions between the recharge water and the rock matrix are

important to the overall project feasibility. Frequently, existing data are available identifying potential problems. Desktop level geochemical modeling or batch testing of the parent rock with recharge water may provide an indication of potential geochemical issues. Lastly, degradation of THMs does occur but still may be problematic due to persistence of some THM components including chloroform. Lastly, periodic back flushing of the ASR well is effective in reversing moderate well clogging. Also, allowing the initial pulse of recharge water to flow to waste can be effective in preventing built up rust and solids from entering the ASR well.

Hilton Head Island ASR site. The Hilton Head Island ASR project is located on State Highway 170 in Okatie, South Carolina, just northwest of Hilton Head Island. The ASR project is located at the Chelsea Water Treatment Plant administered by the Beaufort-Jasper Water and Sewer Authority (CH₂M Hill, 2001). The ASR project is utilized to assist the Authority in meeting its peak summer demands or emergency supplies.

The two ASR test wells (ASR 1 and TW 1) are completed within the Floridan Aquifer System (FAS), a series of marine carbonate aquifers composed of limestone, sandy limestone, and dolomite separated by intermediate confining units composed of clay, sand and low permeability limestone or mudstone (Bush and Johnson, 1988). The ASR wells are completed in the confined Upper Floridan aquifer flow zone. This zone is quite permeable and is characterized by permeable beds of poorly indurated marine skeletal fragments and calcareous sands to highly vuggy limestones and cavernous dolomites (CH₂M Hill, 2001). At the ASR project site, geophysical logs were utilized to evaluate the aquifer during aquifer testing. Caliper logs measuring the borehole diameter

revealed several large open cavities in both wells. These zones were thought to contribute up to 50% of the total flow out of the well. Conversely, during ASR recharge, up to 50% of the water may preferentially flow in these zones. Aquifer testing determined that the average transmissivity is 38,400 ft²/day and the average storage coefficient is 2.44×10^{-4} (CH₂M Hill, 2001).

The source water and ambient groundwater quality is typically excellent at the project site. The treated source water is characterized by low TDS concentrations of less than 72 mg/l, low average TSS of 1.5 mg/l, moderate oxygen content of 8 to 12 mg/l, and a moderately high TOC of 3.5 mg/l. In addition, ammonia and orthophosphate are slightly elevated, while TTHMs are less than regulatory standards with a concentration of 71 mg/l. The groundwater quality is good with a low concentration of TDS and low concentrations of iron, manganese, and sulfate. In addition, the concentration of combined radium and radon are slightly elevated.

Analysis of the recovered water during cycle 1 and cycle 2 revealed minor geochemical reactions including slight dissolution of the calcium carbonate contained in the limestone matrix. Concentrations of TTHMs and HAAs during cycle 1 declined from 23.5 ug/l to 17.2 ug/l and 25.4 ug/l to 14.9 ug/l, respectively, during recovery (CH₂M Hill, 2001). It should be noted that the decline in concentration of HAAs (41% decline) was much greater percentage-wise than the decline in TTHMs (27% decline). Recovered water also exhibited a high color value during the initial recovery period compared to later periods. A similar pattern was noted for turbidity. Cycle 2 test data revealed similar patterns. One other interesting note is that the average concentration of TTHMs measured during cycle 2 recovery was greater than the average value reported for the

recharge water. This may be due to natural variations in the recharge water or in-situ formation of TTHMs at the project site. None of the TTHMs were above regulatory limits at any time. Minor well clogging was identified during recharge operations. Specific injectivity values of the ASR wells declined during recharge operations but were fully restored during periodic well re-development efforts. Recovery efficiency was reported at 67% after cycle 2.

No economic analysis data were available in references reviewed for this project, however, the Hilton Head Island project is expected to be quite economical based upon the shallow depth of the ASR wells and excellent water quality in both the source water and ambient groundwater.

Lessons learned from this site are that well clogging issues are important at most ASR project sites, even those similar to Hilton Head, where the ASR storage zone is highly permeable. As ASR projects are designed for ever-larger inflows, care must be taken to fully evaluate this issue.

Myrtle Beach ASR site. The Myrtle Beach ASR project is located in Horry County, South Carolina adjacent to the Atlantic Intercoastal Waterway (Castro, 1995). Continuing growth in the area caused the City of Myrtle Beach to expand their water supply capability. Currently, a large surface water treatment plant withdrawing water from the Atlantic Intercoastal Waterway provides water supply to the city. The supply is plentiful but the unit cost for customers has increased and the peak treatment capacity is rarely utilized (Castro, 1995). An ASR project has been constructed to provide an alternate means to meet peak water demands at a lower unit cost.

The ASR project is completed within the Black Creek Formation, an aquifer system studied intensively over the last 20 years (Castro and Hockensmith, 1987). The Black Creek Formation is located between 288 and 961 feet below land surface at the test site. The aquifer is a confined aquifer consisting of unconsolidated sediments including fine-grained, micaceous, phosphatic and glauconite-rich sand and clay (Castro and Hockensmith, 1987). The aquifer contains phosphate in the form of shark teeth, pyrite as casts, shell fragments, and thin layers of calcareous sandstone (Castro, 1995). Aquifer testing revealed a transmissivity of approximately 1,600 ft²/day (Castro and Hockensmith, 1987; CH₂M Hill, 1990).

The source water quality is generally excellent from the Atlantic Intercoastal Waterway. It is characterized as containing low concentrations of metals, moderate alkalinity (42.6 mg/l), and moderate dissolved oxygen (6.9 mg/l). It has a high value of TOC of 5.7 mg/l (Castro, 1995). The groundwater is only marginal for potable supply and contains high concentrations of sodium, chloride, alkalinity, and fluoride. Between 1991 and 1992, 10 recharge and recovery tests were completed at the site. The last of these was a long-term cycle test aimed at reviewing a cycle similar to potential future site operations. During each cycle, minor well clogging occurred but was reversed through a regular program of back flushing. Back flushing frequency ranged from daily to bi-weekly. The program was highly successful in maintaining specific injectivity of the ASR well.

During recovery operations for each cycle, data were reviewed and it was determined that mixing of the source water and the ambient groundwater was the dominant process occurring. However, mixing mostly occurred at the edges of the

injected freshwater bubble as shown by the steep cycle one ASR recovery curve in Figure 2-6.

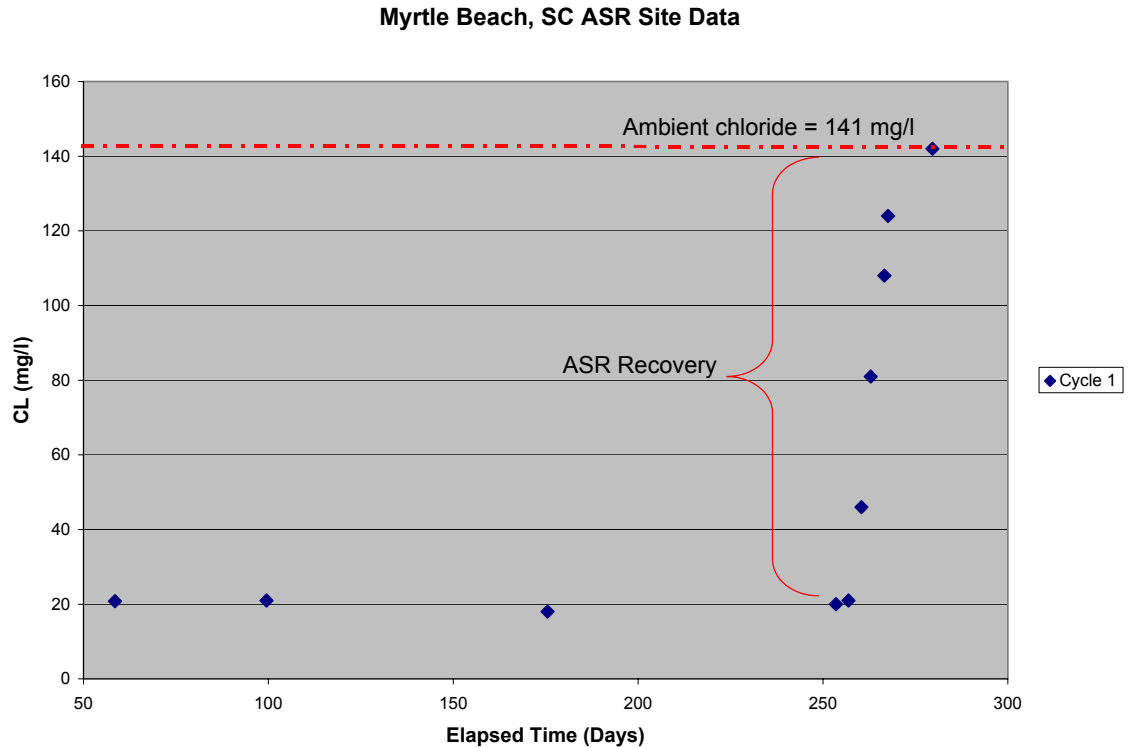


Figure 2-6. ASR recovery curve for cycle 1 at Myrtle Beach, South Carolina

In addition, geochemical reactions were also important for some constituents such as sodium, calcium, bicarbonate, iron, and a few metals including phosphate (Castro, 1995). Based upon geochemical modeling completed for the project study, the most important geochemical reactions were pyrite oxidation, calcite dissolution, and calcium-sodium exchange (Castro, 1995). Castro (1995) also found that close to the ASR well, aerobic reactions dominated the geochemistry while further away, anaerobic reactions were the norm.

An economic analysis completed by the City of Myrtle Beach estimated long-term costs of the project to be approximately \$0.48 per 1,000 gallons recovered using an existing well and \$0.97 with a newly constructed well (Castro, 1995). Castro (1995) also

clearly shows that the unit costs are not completely linear when compared against percent volume recovered or recovery efficiency. As the percent of the recharged water that can be recovered drops, the unit price rises rapidly. For instance, at 10% recovery efficiency, the unit cost per thousand gallons is almost 11 times higher than at 100% recovery efficiency. However, at the 100% recovery efficiency value, the cost of a long-term ASR project is still half as much as a plant expansion of the source water treatment facility.

An important lesson learned from this site concerns ASR economics. Care must be taken in the economic evaluation to truly determine the expected well recovery efficiency as this could greatly affect project feasibility.

Wildwood ASR site. The Wildwood ASR site is located along the southeastern New Jersey coast on the Cape May peninsula of New Jersey (Pyne, 1995). Wildwood is a resort community inundated with summer tourists each year. This tourist influx leads to substantial peak summer water demands that must be met. Water supply is derived from the Rio Grande well field located five miles inland. The well field has 13 operating wells with a combined capacity of 13.5 MGD. In order to meet rising summer water demands, the City of Wildwood began recharging two of the wells during off peak months (Pyne, 1995). Two additional ASR wells were added to the system in the early 1980s. Every year 80% of the water recharged into the wells is recovered to meet peak water demands. This site has been identified as the first ASR site to be constructed within the United States (Pyne, 1995).

The four ASR wells are screened within the Cohansey Aquifer that is an unconfined aquifer composed of marine sands and gravels interbedded with clays. At the site the transmissivity has been estimated to be 11,600 ft²/day (Pyne, 1995).

An economic evaluation of the system completed by the author has estimated the unit cost of the project to be very low ranging from \$0.05 to \$0.10 per thousand gallons recovered. The estimate is considered very crude given the limited information available in the literature. This appears to be a very cost effective project with the low cost largely due to the minimal water treatment required at the site along with the large amount of water that can be stored at the site for later recovery.

The most important lesson learned at the site is the importance of periodic well re-development required for most ASR projects, especially those constructed within unconsolidated sediments. The Wildwood project backflushes the ASR wells daily for 10 minutes to remove rust and iron floc formed in the aquifer (Pyne, 1995). The wells are also completely re-developed and cleaned every five years to restore full well capacity (Pyne, 1995). This program has been ongoing for approximately 35 years and has been extremely successful.

Lychett Minster. The Lychett Minster ASR project is located in England along the southeast coast east of Dorchester, England (Eastwood and Stanfield, 2001). The project was conceived as a pilot project in the Chalk Aquifer to evaluate the feasibility of ASR in the region.

The Chalk Aquifer is a marine carbonate aquifer that has a dual-porosity nature (Gaus et al., 2000). Figure 2-7 displays a hydrogeological cross-section of the project area (Stanfield, 2001). The site has been discussed frequently in the literature as a notable ASR site in that it has been considered an unsuccessful project.

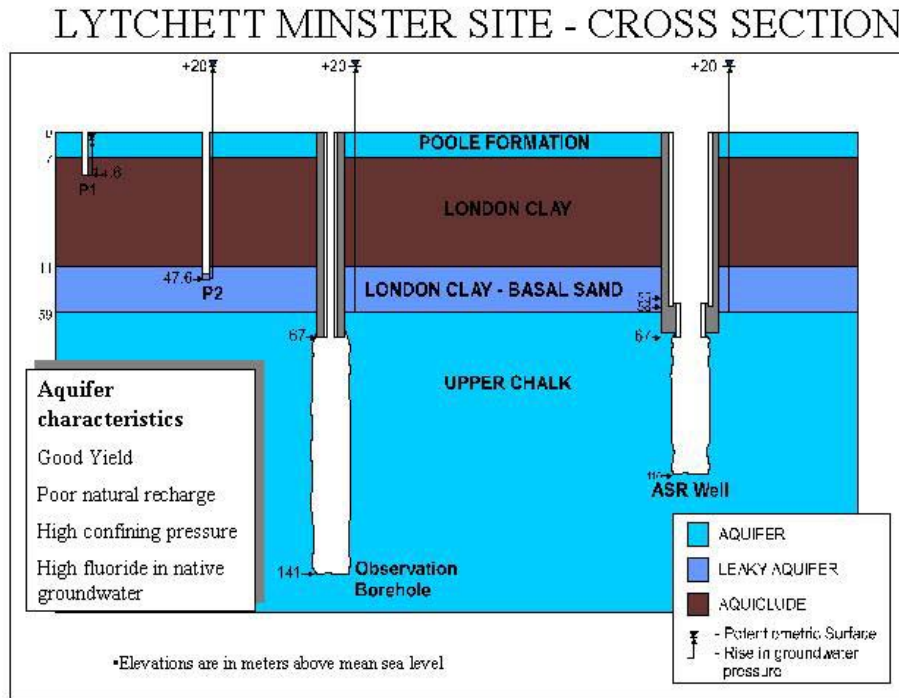


Figure 2-7. General hydrogeological cross-section of the Lytchett Minster site in England (used with permission from Paul Stanfield, Wessex Water)

Aquifer testing has revealed an average transmissivity of 2,150 ft²/day. At the project study area, the overlying London Clay confining unit confines the Chalk Aquifer.

The water quality of the treated source water was excellent with low TSS, TOC, and metals. The ambient groundwater on the other hand, contained objectionable concentrations of fluoride in excess of promulgated standards (Eastwood and Stanfield, 2001). Multiple recharge and recovery cycles were completed at the site, however, even after nine cycles, the concentration of fluoride in the recovered water exceeded regulatory standards. Similar to other ASR projects, the recovered water quality improved over subsequent cycles but seemed to reach a steady-state concentration after approximately four cycles, after which, minimal improvements were noticed. Figure 2-8 shows the

fluoride results after four ASR cycles compared to other literature results that plotted TDS.

LYTCHETT MINSTER RESULTS COMPARED TO TYPICAL RECOVERY QUALITY CURVES (PYNE 1995)

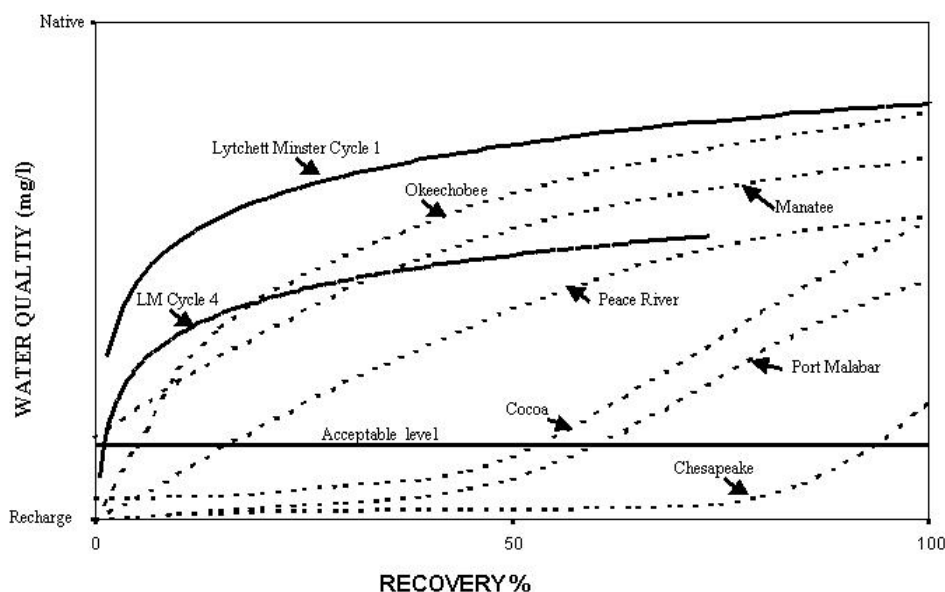


Figure 2-8. Cycle testing results compared to other published data from Pyne (1995) (used with permission from Paul Stanfield, Wessex Water)

Due to the water quality issues at the site, the pilot project was ultimately abandoned. Modeling completed after the project indicated that a combination of dual-porosity behavior during mixing and geochemical reactions probably led to the water quality problems (Gaus et al. 2000; Gaus, 2001). Dissolution of the mineral fluorite within the preferential flow zones was thought to have contributed to the water quality issues.

An economic evaluation of the project was not available within the literature reviewed, however, if the fluoride problem could have been resolved, the project appeared to be quite promising from an economic standpoint.

The main lesson learned in this project is that dual-porosity aquifers probably require more water quality conditioning than porous-media type aquifers (Gale, 2002). In

addition, in these types of aquifers, diffusion may be a dominant process as compared to hydrodynamic dispersion.

Windhoek ASR site. The Windhoek ASR project is located in southwestern Africa in central Namibia. The area is semi-arid with precipitation of only 8 to 15 inches per year (Murray, 2004). While the aquifer only supplies 10% of the city's water supply, it has become severely over drafted, providing an opportunity for a large ASR system (Murray, 2004).

The geology of the area is composed of mainly schist and quartzite. Intense faulting/folding has resulted in a highly fractured aquifer system (Murray, 2004). Borehole injection tests using city treated domestic water were run. The longest test lasted 195 days with the highest injection rate achieved being 1.36 MGD. Transmissivity data for the Windhoek Aquifer has been estimated to range from 650 ft²/day for the formation as a whole to 43,000 ft²/day for major fractured areas (Murray and Tredoux, 2002). During injection tests using low turbidity water (less than 1 NTU) and water low in dissolved organic carbon (2.0 mg/l), no major well clogging issues were reported (Murray and Tredoux, 2002).

Limited data are available for this site currently, however, the ASR system for the City of Windhoek is planned for further expansion to seven ASR wells that will be able to provide for 87% of the city water supply needs (Murray, 2004). The lesson learned here is that ASR can be successful in a wide range of hydrogeologic environments, even highly fractured bedrock, even though the recovery efficiencies in such aquifers could be low. Similar findings have been reported from several study sites in Australia (Harrington et al., 2002). They conclude that recovery efficiencies in fractured rock

systems may be low due to the fact that any injected water will rapidly mix with the native groundwater and be transported away from the ASR well due to high groundwater velocities that exist in rock fractures.

Brackish water ASR sites

Thirty (30) brackish water ASR sites were reviewed for this report. Of the 30 sites surveyed, 21 of them are located in the State of Florida within the United States. Of the remaining nine sites, four are located in the United States and five are international sites. The 30 sites represent diverse geologic environments as well as different operating types. Figure 2-9 shows the location of the brackish water ASR sites for this report.

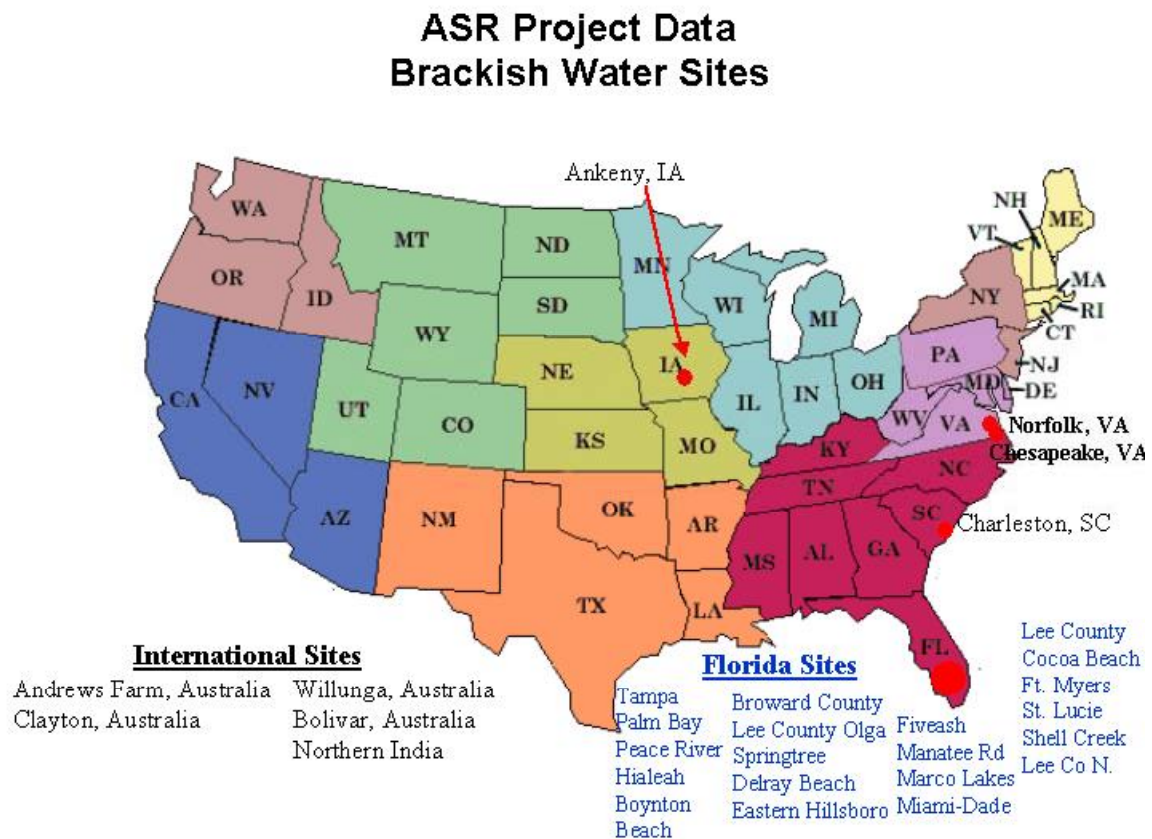


Figure 2-9. Location of brackish water ASR project sites where data were collected for this report.

Available data ranged from extensive reporting for Ankeny, Andrews Farm, Boynton Beach, Cocoa Beach, Palm Bay, Hialeah, Tampa, Peace River, Charleston, Willunga and Norfolk, to short summaries available for Lee County, Fort Myers, Northern India, Bolivar, Manatee Lake and Clayton. Electronic data were provided for the Boynton Beach, Lee County North, Lee County Olga, Tampa, Palm Bay, Cocoa Beach, Fiveash, and Marco Lakes sites. These data facilitated development of unit water costs for a few of these sites. A number of the Florida sites have been lumped together for the sake of brevity due to similarities in hydrogeology and ASR operations.

Ankeny ASR site. The Ankeny ASR project is located in eastern Iowa near the City of Des Moines, Iowa (Miller et al. 1998). A large-scale ASR demonstration project was completed between 1995 and 1998 including approximately one year of recharge and recovery cycle testing (Millet al. 1998). The project sponsor was the Des Moines Water Works (DMWW) and was prompted by catastrophic flooding experienced by the utility in 1993 during large Mississippi River and tributary flooding. Due to the flooding, the DMWW was unable to provide water supply to 250,000 customers for 11 days. The ASR project was envisioned to provide a reliable emergency water supply (Miller et al. 1998).

The goal of the ASR system is to provide 30 MGD of water supply capacity in order to meet 90-days worth of emergency demand. An analysis of available data indicated that typically peak demand occurs during the summer season from July to August (Miller et al. 1998). The testing program was able to modify two existing water supply wells formerly used in the Ankeny, Iowa area with the DMWW service area. The wells are completed within the Jordan Aquifer, part of the regional Cambrian-Ordovician

Aquifer (Miller et al. 1998). The aquifer is composed of sandstone and dolomite flow zones, with the dolomite zones dominant. The wells are approximately 2,700 feet deep with an open-hole interval from 2,100 to 2,700 feet. Aquifer test results indicate a confined aquifer with a transmissivity of 13,800 ft²/day and a storage coefficient of 2.7×10^{-5} as compared to regional values of 5,000 ft²/day and 2.5×10^{-4} , respectively (Miller et al. 1998; Young, 1992).

Three ASR test cycles were completed with the third cycle continuing for the longest duration. The treated source water used for recharge activities was of excellent quality, meeting all regulatory standards with a TDS of 200 mg/l, TSS of less than 2.5 mg/l, TOC of 1.0 to 2.0 mg/l, and dissolved oxygen of 3 to 13 mg/l (Miller et al. 1998). The chlorine residual ranged from .20 to .70 mg/l. The ambient groundwater is considered to have poor quality with high TDS values of 750 mg/l, high fluoride, iron, sulfate, and radionuclide (combined radium) concentrations (Miller et al. 1998). In addition, during cycle three recharge operations, the temperature of the recharge water was only 9 degrees Celsius whereas the ambient groundwater had an average temperature of 25 degrees Celsius. This substantial temperature difference undoubtedly lowered the “apparent transmissivity” of the aquifer due to temperature viscosity effects. The lower apparent transmissivity led to higher than expected aquifer head changes during the cycle three testing program. During the testing program only minor well plugging was observed. This small decrease in well efficiency was easily rectified through periodic back flushing. Minimal geochemical reactions were noted at the site with the exception of ion exchange of calcium for sodium and minor dissolution. Limited data were collected concerning DBPs but data that were collected indicated that the TTHMs

recovered were generally less than 5 ug/l versus an average recharge concentration of 22 ug/l. ASR recovery efficiency of the site was recorded at approximately 70% during cycle 3 due to regulatory exceedance of TDS and combined radium in the recovered water (Miller et al. 1998). Further cycles are expected to reach recovery of close to 100% of the recharged water.

An economic evaluation completed by DMWW estimated the unit cost of a full-scale system to range from \$0.44 to \$0.63 per thousand gallons recovered (Miller et al. 1998). It was also noted that the final unit price would be very sensitive to the amount of water recovered from the aquifer. In this case, however, the actual amount of water recovered is not as important as being able to provide a dependable emergency water supply for 90 days.

Several important lessons can be gleaned from this project. First, the project is an excellent example of the use of ASR to meet short-term emergency demands. Second, ASR can be successful in a deep brackish aquifer as long as good quality source water is available.

Charleston ASR site. The Charleston ASR site is located near the City of Charleston, South Carolina. The ASR project was originally conceived as an emergency municipal drinking water supply for the southern portion of the city which is subject to flooding in the event of a hurricane, or a water main break during a hard freeze or possible earthquake (Mirecki et al. 1998). Water supply for the pilot ASR project was treated Edisto River water.

The ASR test well was completed within the Santee Limestone/Black Mingo confined aquifer. The aquifer is characterized by carbonate limestone, fracture-

dominated sandstone, and interlayered crystalline limestone (Campbell et al. 1997). The aquifer properties have been determined through analytical and numerical modeling. The transmissivity of the aquifer is estimated to range from 130 to 220 ft²/day while the storage coefficient ranges from 1.0×10^{-4} to 5.5×10^{-4} (Campbell et al. 1997).

The treated source water is generally of excellent quality and includes low concentrations of chloride (11 mg/l), calcium (18 mg/l), and sulfate (36 mg/l). The source water also contained an average of 104 ug/l of TTHMs, low dissolved oxygen (3.0 mg/l), moderate TOC (2.8 mg/l), and moderate amounts of TSS (1.0 to 3.0 mg/l). The groundwater is brackish and includes objectionable concentrations of chloride (1,700 mg/l), sodium (1,300 mg/l) and sulfate (220 mg/l) (Mirecki et al. 1998). During ASR cycle recovery, several observations are apparent. First, TTHM concentrations generally decreased at the ASR well as recovery progressed, however, concentrations sometimes exceeded the regulatory limit of 100 ug/l prevalent at time. Second, aquifer testing data throughout the cycle testing program revealed that the aquifer transmissivity actually increased likely due to minor dissolution of the carbonate matrix. In addition, minor upconing of highly brackish groundwater occurred during the last few of the 13 recovery cycles. Lastly, recovery efficiency of the recharged water generally improved in subsequent cycles from 38% to 61% with a few cycles possibly affected by upconing (Campbell et al. 1997).

In-depth economic evaluations were not available in any of the references reviewed related to the Charleston project, however, the project should be feasible for emergency water storage.

A few important lessons can be derived from a review of this project. First, the use of ASR for emergency water supply seems like an appropriate use of the technology. Second, brackish water ASR sites can be subject to upconing of highly saline water during ASR recovery operations. Subsurface investigations should thoroughly define the geology and water quality in zones below the ASR storage zone.

Norfolk ASR site. The Norfolk ASR project was a large-scale test project completed in the early 1970s. During late 1971 and early 1972, four recharge and recovery cycles were completed adjacent to a Norfolk, Virginia water treatment plant (Brown and Silvey, 1977). During this time, excess freshwater from surface water reservoirs was available during low peak winter times. The United States Geological Survey and the City of Norfolk cooperated on a study to store the water in a brackish water aquifer (Brown and Silvey, 1977). A deep ASR well was constructed with a screened interval from 891 to 973 feet below ground surface.

The hydrogeology in the area is composed of unconsolidated sand, gravels, and clays of the Atlantic Coastal Plain physiographic province (Brown, 1971). The aquifers are of marine origin and contain wood fragments, ostracodes, and foraminifers, suggesting a littoral environment (Brown, 1971). Laboratory testing revealed that the sands had a medium particle size diameter of 0.20 millimeters indicating predominately medium sand in the aquifer. The aquifer storage zone was also bounded by clay units, rich in illite and montmorillonite (Brown and Silvey, 1977). Clays of this type are moderately to highly sensitive to water quality and may result in swelling or dispersion. Aquifer testing of the confined storage zone and regional modeling of the same estimated the transmissivity to range from 5,360 to 16,600 ft²/day with a storage coefficient of 1.5 x

10^{-4} (Brown and Silvey, 1977). Geophysical flow logging indicated that up to 40% of the groundwater flow into the well was concentrated between the 900 to 920 feet deep interval, the remaining 60% emanating from 920 to the screen bottom.

The water quality of the source water was excellent, containing low concentrations of TSS (1.0 to 2.0 mg/l), low TDS (111 mg/l), low chloride (21 mg/l), and moderate dissolved oxygen of 8 to 11 mg/l (Brown and Silvey, 1977). The groundwater was moderately brackish containing a TDS concentration of 3,000 mg/l, chloride concentration of 1,360 mg/l, and sodium concentration of 1,140 mg/l. During recharge and recovery operations, geochemical reactions were observed by plotting the concentration of various constituents (e.g., calcium, magnesium and sodium) during recovery versus chloride concentrations during recovery. The review of this data indicated that recovered water concentrations of these substances were higher than either the source or ambient concentrations, demonstrating some type of geochemical reaction was happening. In addition, as the recharge and recovery tests progressed, the ASR well specific capacity was continually reduced. Some of the reduction was due to well clogging with TSS, however, a majority was attributed to dispersion of the aquifer clays due to cation exchange reaction between calcium and sodium in the mixed water (Brown and Silvey, 1977). The clay dispersion led to systematic reductions in the aquifer hydraulic conductivity distribution. The resulting conductivity distribution was highly heterogeneous and reduced the ASR recovery efficiency from 65% in recharge/recovery tests # 1 and 2 to 20% in test # 4 (Brown and Silvey, 1977).

No economic evaluations were completed for this test project, however, the conclusion was that storing freshwater in the aquifer could be feasible if the clay dispersion issues could be solved using aquifer chemical pre-treatment.

The primary lesson learned at this site is that clay dispersion is a real problem that could lead to well or aquifer clogging adjacent to the ASR well. The clogging could severely limit the ASR effectiveness and require significant remedial actions to fix the problem. ASR storage zones containing sensitive illite or montmorillinite clays should be evaluated carefully for clay dispersion potential. In addition, a preliminary evaluation of freshwater buoyancy stratification indicated little, if any, stratification occurred at the test project (Brown and Silvey, 1977). Brown and Silvey (1977) suggest that natural aquifer stratification may have prevented the upward migration of freshwater during ASR storage.

Chesapeake ASR site. The Chesapeake ASR project is located in Chesapeake, Virginia along the Northwest River. The area has experienced tremendous population growth and was expected to double its average and peak water demands by the year 2030 (Dwarkanath and Ibison, 1991). In addition, the current source of freshwater is the Northwest River. This coastal river is subject to saltwater intrusion during dry periods resulting in chloride levels that exceed the regulatory level (Dwarkanath and Ibison, 1991). Therefore, ASR was selected to provide a dependable seasonal water supply to augment the Northwest River source. Construction of the pilot ASR facilities began in 1988 and three cycles of ASR recharge, storage and recovery began soon after construction was complete. The ASR well was constructed of epoxy-coated steel to a depth of 725 feet below ground surface and contains a vertical turbine pump (Pyne,

1995). By the end of the third cycle test, it became evident that the ASR project would be highly successful (Dwarkanath and Ibison, 1991). Operations completed after pilot testing were successful in storing 350 million gallons of freshwater for seasonal use.

The ASR storage zone includes permeable sections of the Upper and Middle Potomac aquifers of Cretaceous Age (Pyne, 1995). The aquifer is a confined system and is characterized by interbedded sands and clays of marine origin. Aquifer performance tests have determined that the average transmissivity of aquifer at the ASR project site is 9,810 to 12,370 ft²/day with a storage coefficient of 4.5×10^{-5} (Pyne, 1995). Cycle testing recharged 5, 24, and 88 million gallons of treated water, respectively, during three test cycles.

The water quality of the treated source water was fair to good with TDS concentrations of 438 mg/l, TSS concentration up to 6.0 mg/l, low coliform counts, and TTHMs of 109 ug/l (Pyne, 1995). The ambient groundwater contained TDS values of 500 to 1,000 mg/l, chloride concentration of 280 mg/l, sodium concentration of 359 mg/l, and moderate concentrations of manganese and iron (Pyne, 1995). During pilot testing, no major geochemical issues were observed and well clogging was moderate requiring periodic back flushing. During later ASR operations, the pH of the recharge water was lower than it had been during testing. The low pH leached iron and manganese out of the aquifer and resulted in a secondary water quality problem in the recovered water. The solution to this geochemical issue was the simple control of the recharge pH. The well clogging was attributed to the higher TSS values resulting from high treated-water color and alum concentrations (Pyne, 1995). After three cycles, almost 100% of the recharged water could be recovered before exceeding any regulatory threshold.

Economic evaluations of the project indicated that the ASR option was significantly less expensive than other water supply options, however, unit costs were not available in the literature reviewed for this report.

The primary lesson learned from this project is that pH control of the recharged water may be important in some aquifer types that contain minerals sensitive to pH. Generally, pH control is an inexpensive treatment option at any project site.

Boynton Beach ASR site. The Boynton Beach ASR project is located in the southeastern coast of Florida at the City of Boynton Beach's water treatment plant. The Boynton Beach water treatment plant generally is subjected to high summer water demands (CH₂M Hill, 1993). The original ASR facility included one ASR well with a vertical turbine pump and other associated site infrastructure. The ASR well is completed in the lower Hawthorn Formation between 800 and 900 feet below ground surface (Pyne, 1995).

The ASR storage zone is within the confined lower Hawthorn Formation that has been loosely connected with the Floridan Aquifer System (Miller, 1997) and is characterized as a sandy marine limestone. Aquifer performance testing has indicated an average transmissivity at the project site of 9,358 ft²/day (CH₂M Hill, 1993). Regional storage coefficients of this zone range from 1×10^{-3} to 1×10^{-5} (Miller, 1997). During well drilling and development, large quantities of fine sand were pumped from the well during airlift development (Pyne, 1995).

The recharge source water quality was fair to good and was characterized by low chloride concentrations (49 to 55 mg/l), low TDS values of (156 to 178 mg/l), high TSS up to 23 mg/l, high TOC (12 to 27 mg/l), high TTHMs (152 to 170 mg/l), and moderate

values of free chlorine ranging from 2.5 to 4.0 mg/l (CH₂M Hill, 1993). The ambient groundwater was characterized by high chloride concentrations (1,920 mg/l), high TDS (3,910 mg/l), and low ammonia (0.73 mg/l). Recovery of the recharged freshwater has been highly successful at the project. As of 2002, 16 separate recharge and recovery cycles have been completed with recovery efficiencies varying from 40% to over 100% (Reese, 2002). Recovery efficiencies greater than 100% are possible at the site since recovery operations frequently recover water beyond the 250 mg/l chloride regulatory limit (e.g., cycle 1 recovered to a chloride concentration of 756 mg/l). Figure 2-10 depicts the results of Boynton Beach cycle 1 recharge and recovery.

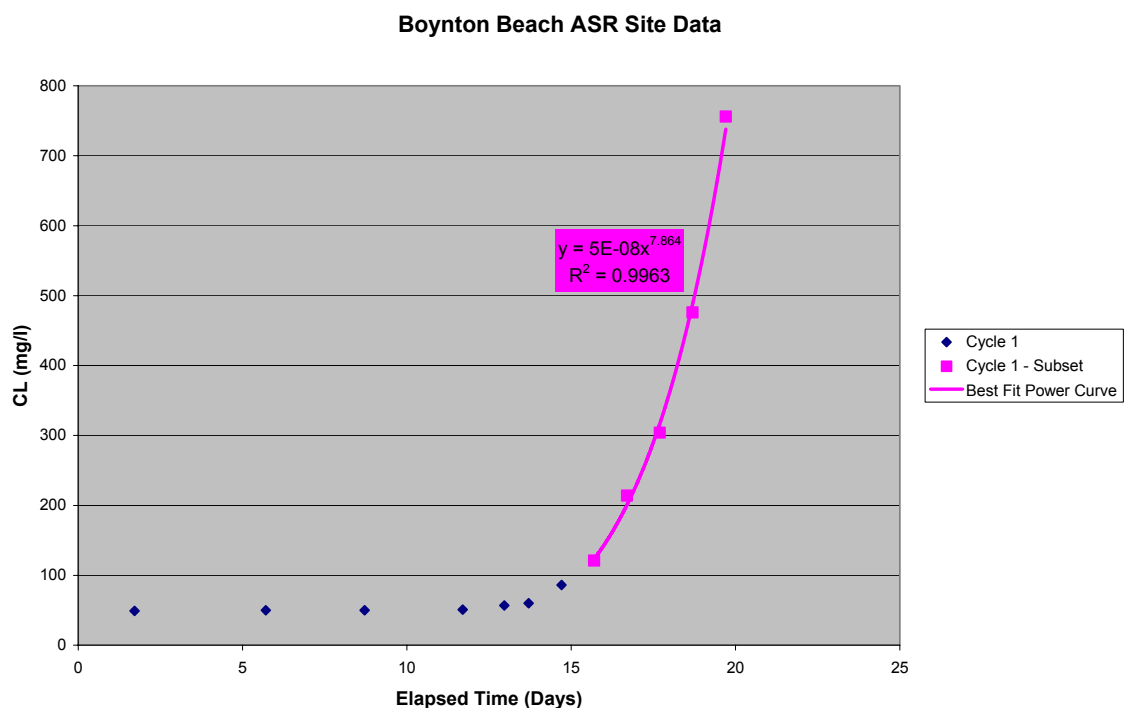


Figure 2-10. Boynton Beach cycle 1 recharge and recovery cycle chloride water quality versus elapsed time in days

Based upon the best-fit power curve applied to the chloride recovery data, the recovery efficiency of cycle 1 was estimated to be 47% (as defined by a chloride limit of 250 mg/l). The Boynton Beach data also showed that over multiple cycles, the ASR

cumulative recovery efficiency improves due to residual freshwater left in storage after each recharge and recovery cycle. Figure 2-11 shows the cumulative recovery efficiency, as defined previously, versus cycle number at Boynton Beach, along with a long-term trend line.

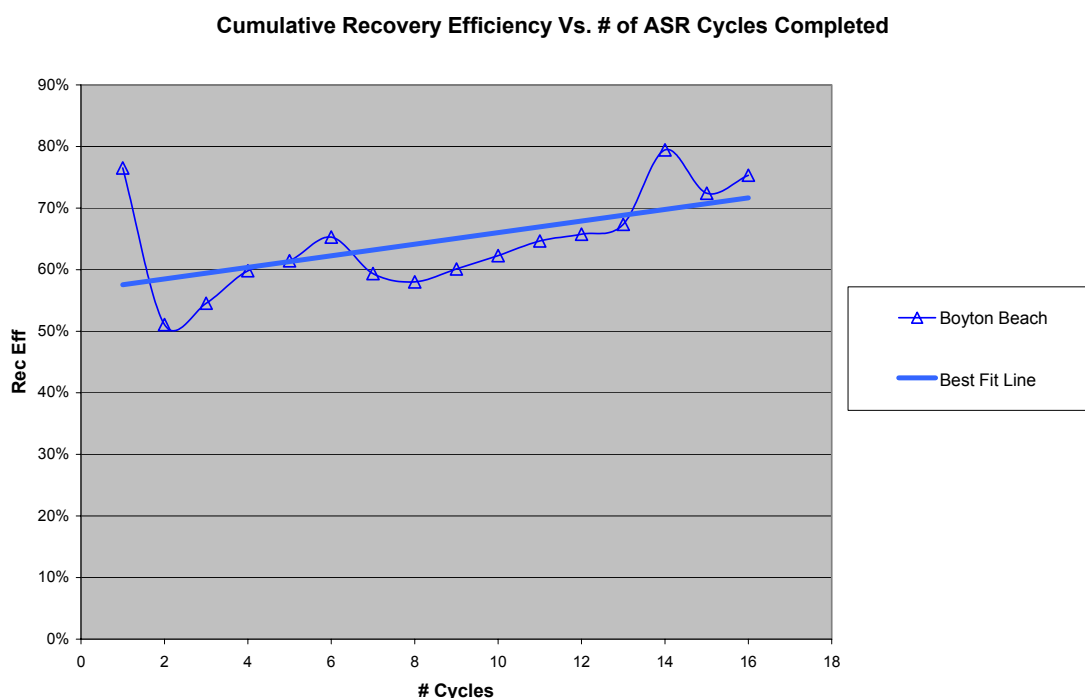


Figure 2-11. Boynton Beach long-term cumulative recovery efficiency versus ASR cycle number along with trend line for same.

Recovered water during cycle 1 was noted to contain TTHM levels higher than the maximum concentration observed during recharge operations (Pyne, 1995). This possibly indicates that THM formation occurred in-situ due to high TOC and residual chlorine values contained in the recharge water. Periodic back flushing at the beginning of a recovery cycle has been sufficient to maintain original recharge specific capacities (Pyne, 1995).

A preliminary economic evaluation completed in 1993 indicates that the project was considerably less expensive than other seasonal water supply options (CH₂M Hill, 1993;

Muniz and Ziegler, 1995). Muniz and Ziegler (1995) report that ASR is usually more than 50% cheaper than other options, such as surface storage reservoirs. An economic evaluation of the Boynton Beach ASR project has been completed for this report by the author based upon information contained within the references reviewed as well as standard water treatment and electricity costs. The unit cost of the recovered water is estimated to range from \$1.79 to \$2.23 per thousand gallons recovered. It was also noted that the final unit price would be very sensitive to the amount of water recovered from the aquifer on an annual basis. Also, it should be noted that the ASR system has been successfully utilized since 1993 and has always been available in the event of an emergency.

The primary lesson learned from this project is that ASR can be successful in a brackish limestone storage zone. The ASR project has provided a reliable water supply option to meet seasonal or emergency demands (e.g., during hurricanes) in southeast Florida (Muniz and Ziegler, 1995). The project also clearly shows the improvement in ASR recovery efficiency over the long term as freshwater is left in a buffer adjacent to the ASR well. High TOC and residual chlorine levels can combine to form additional TTHMs in-situ. Lastly, although TSS and TOC concentrations were high in the recharge water, well redevelopment activities in a limestone storage zone are less onerous than other geologic environments such as in unconsolidated sediments.

Lee County ASR site. The Lee County ASR project is located in southwest Florida along the south side of the Caloosahatchee River, one mile east of the S-79 water control structure (Fitzpatrick, 1986). The Lee County ASR trial was a pilot project conducted between Lee County, South Florida Water Management District and the

United States Geological Survey. The project purpose was to test the feasibility of storing freshwater in a confined, saline aquifer (Fitzpatrick, 1986).

The ASR storage zone selected for the test was the leaky-confined lower Hawthorn Aquifer that is loosely connected to the Floridan Aquifer System (Miller, 1997). The aquifer is a sandy limestone aquifer of marine origin. Based upon aquifer performance testing, the aquifer transmissivity was estimated to be 800 ft²/day with a storage coefficient of 1×10^{-4} (Reese, 2002). The pilot trial consisted of three recharge, storage and recovery cycles. The third cycle was the largest consisting of 8,548,000 gallons of recharged treated water and 20,478,000 gallons of recharged untreated water (Fitzpatrick, 1986). Each of the three cycles had different storage durations with 0, 47, and 98 days respectively.

The source water quality was variable during the three cycle tests. The influent chloride concentration was generally less than 270 mg/l with an average of 96 mg/l and a minimum of 46 mg/l (Fitzpatrick, 1986). Algae were also noted as a potential problematic constituent. In addition, the source water contained moderate concentrations of fluoride (0.1 to 0.6 mg/l), arsenic (1 to 5 ug/l), and iron (0.12 to 0.37 mg/l). Data contained within Fitzpatrick (1986), reveal that the ambient groundwater was brackish with 500 to 550 mg/l chloride and a TDS value of 1,580 mg/l. Sulfate in the ambient groundwater was also elevated with a concentration of 340 mg/l (Fitzpatrick, 1986). Moderate concentrations of heavy metals were also detected in the groundwater including fluoride (1.6 mg/l), arsenic (1 ug/l), and strontium (15 mg/l). Temperature differences between the recharge water and the ambient groundwater were also noted and taken into account during data evaluations.

During recharge operations, severe well clogging was noted in cycle one due to the high recharge TSS concentration that ranged from 7.0 to 40.0 mg/l in the untreated source water (Fitzpatrick, 1986). Head buildup due to clogging reached 193 feet after 37 minutes of injection. The recovery efficiency of cycle one was estimated to be 38.7% before the chloride concentration exceeded 250 mg/l during recovery. The initial quality of the cycle one recovered water was poor and contained high TSS, iron, phosphorus, nitrate, and color. After this test, the ASR well was back flushed and acidized to improve the well specific capacity. During recovery operations of cycle 2, the water quality was also poor with the chloride level exceeding 250 mg/l after only 9.7% of the recharged water was recovered. Bacteriological data of the recovered water indicated that there was marked bacterial growth at the ASR well and at the nearest observation well (Fitzpatrick, 1986). The bacterial growth may have contributed to uneven recovery of the recharged water, leading to poor ASR performance for cycle 2. In addition, the recovered water contained elevated concentrations of iron and arsenic, possibly indicating an ongoing geochemical reaction in the aquifer. Results from cycle 3 were generally similar with the exception that the chloride concentrations rose at a slower pace and resulted in a recovery efficiency of 30.4% (Fitzpatrick, 1986). Also, bacteria typing of recovered water revealed that no methanogenic bacteria or denitrifiers were found, possibly due to recharge using treated drinking water for a portion of cycle 3. The number of anaerobic bacteria recorded during cycle 3 recovery was similar to cycle 2 (Fitzpatrick, 1986).

No economic evaluations were completed for this project but it should be noted that additional ASR wells have been installed at this facility by Lee County (Reese, 2002).

Therefore, Lee County has indicated the project is feasible based upon continued financial investment in ASR at the water treatment plant.

There are multiple lessons to be learned from this ASR project. First, high concentrations of TSS in the recharge water can lead to severe well clogging in a moderately permeable aquifer, even if the storage zone is limestone. Second, recharge with high TSS can result in growth of bacteria in the aquifer and adjacent to the ASR well, resulting in poor system performance. Third, water viscosity differences due to recharge and ambient water temperature differences are important during the interpretation of the head buildup or well drawdown. Lastly, the ASR recovery efficiency generally increases over time as the residual freshwater left in the aquifer forms a buffer, however, long storage times similar to the one utilized during cycle 3, can result in performance reduction. The reduction in the recovery efficiency may be due to movement of the freshwater bubble down gradient, buoyancy stratification effects, or well clogging leading to preferential flow pathways adjacent to the ASR well.

Palm Bay ASR site. The Palm Bay ASR project is located in southeast Florida at the Port Malabar (Palm Bay) water treatment plant. An ASR well has been in operation at this location since 1988 (CH₂M Hill, 1988). The project was undertaken in order to provide the Palm Bay area with a “stopgap” water supply by using ASR to augment the small water treatment plant, while a municipal well field was expanded (Pyne, 1995). It is currently used for seasonal and emergency storage.

The ASR well is completed in a carbonate, sandy limestone semi-confined brackish aquifer zone that is located within the Upper Floridan Aquifer System (CH₂M Hill, 1988). The storage zone is also characterized as moderately leaky. The ASR well is 370

feet below ground surface and has an open-hole interval from 298 to 370 feet (Pyne, 1995). Aquifer testing at the ASR well estimated the transmissivity to be 2,300 ft²/day with a storage coefficient of 1.5×10^{-4} (CH₂M Hill, 1988).

The source water quality is considered excellent and contains moderate levels of chloride (180 mg/l) and TSS (1.5 mg/l), while the ambient groundwater quality is considered poor. The ambient groundwater is brackish and contains chloride concentrations of 588 mg/l, TDS of 1,360 mg/l, sulfate of 125 mg/l, and total hardness of 600 mg/l (CH₂M Hill, 1988). During recovery operations of three different ASR cycles, the recovered water was typically a blend of the recharged and ambient waters. No geochemical reactions were noted, however, elevated arsenic of 8 ug/l was recorded in the recovered water during cycle 3. This concentration is higher than either the recharge or ambient concentration of arsenic, indicating possible geochemical reactions within the aquifer during ASR storage. In addition, cycle 3 recovery data only recorded two THM components, namely, chloroform and dichlorobromomethane (CH₂M Hill, 1988). These compounds seem more persistent within the aquifer storage zone. The recovery efficiency of the Palm Bay ASR project has been excellent with the first cycle approaching 50%. Today, the individual cycle recovery efficiency routinely approaches 100%. The long-term cumulative recovery efficiency is also high as shown on Figure 2-12. An economic evaluation of the Palm Bay ASR project has been completed for this report by the author based upon information contained within the references reviewed as well as standard water treatment and electricity costs. The unit cost of the recovered water is estimated to range from \$2.09 to \$2.62 per thousand gallons recovered. It was also noted that the final unit price would be very sensitive to the amount of water

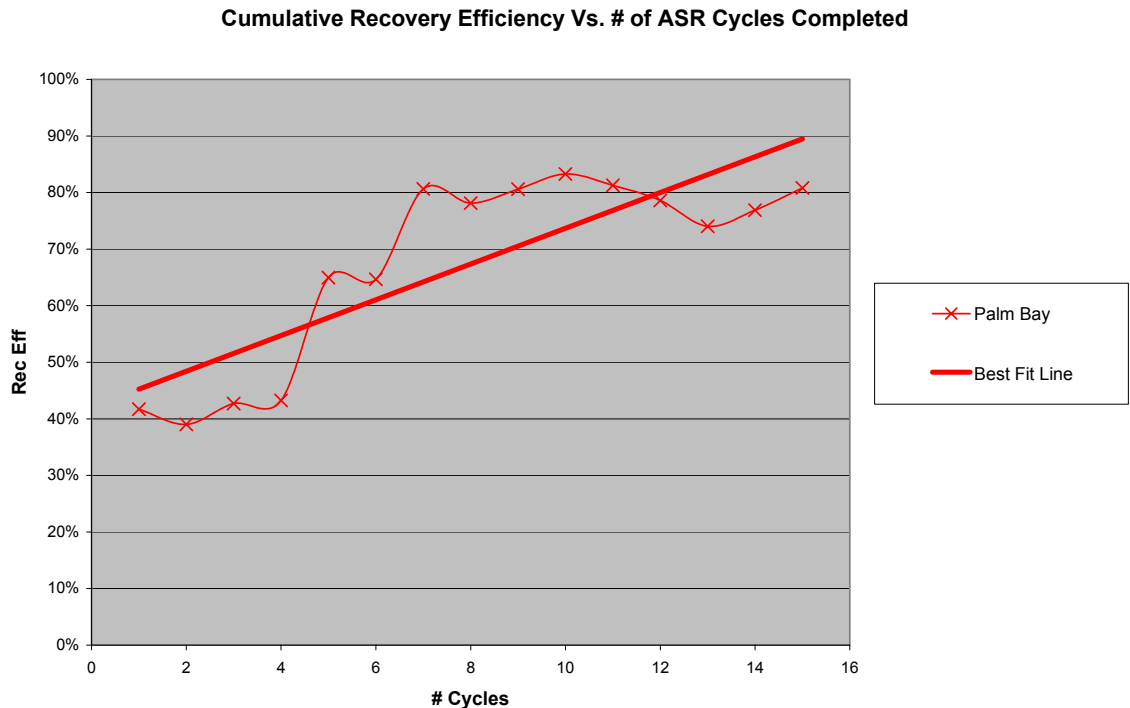


Figure 2-12. Palm Bay long-term cumulative recovery efficiency versus ASR cycle number along with trend line for same (Data include pilot cycle testing as part of cycle 1).

recovered from the aquifer on an annual basis. Also, it should be noted that the ASR system has been successfully utilized since 1989 and has always been available in the event of an emergency. Lastly, the unit cost calculated for the Palm Bay ASR project could be lower if not for the impact of heavy industrial pumping of the Upper Floridan Aquifer. This heavy pumping caused freshwater stored early in the cycle testing program to be pulled away from the ASR well where some of the water could not then be recovered (Pyne, 1995).

A few lessons learned can be derived from a review of the Palm Bay ASR project. First, due to the projects moderate size (less than one MGD), the unit cost of the recovered water is high compared to other similar ASR sites. Second, local groundwater users may impact the distribution of the stored freshwater “bubble” unless provisions are

taken to legally protect the water or ensure the site selection process for the ASR site considers the impacts to and from local well users.

Hialeah ASR site. The Hialeah ASR project is located in southeastern Florida near Miami. The Hialeah ASR project was a test trial of freshwater storage within the brackish Upper Floridan Aquifer System. The test trial was administered by the United States Geological Survey and was conducted between 1974 and 1980 (Merritt, 1997).

The ASR well at the Hialeah project was completed from 950 to 1,100 feet below land surface. A major flow zone, identified through geophysical flow logging, was observed between 1,015 and 1,050 feet (Merritt, 1997). The confined storage interval is characterized as containing shelly limestone of the Upper Floridan Aquifer System. Aquifer performance tests estimated the transmissivity to be 9,600 ft²/day with a storage coefficient of 7.8×10^{-5} (Merritt, 1997; Reese, 2002). Numerical modeling of the cycle tests estimated an aquifer dispersivity of 65 feet.

The ASR cycle testing consisted of three separate cycles, each increasing in volume injected and duration. The storage period was also increased after each recharge cycle. Table 2-2 summarizes the cycle testing data for the Hialeah project conducted from 1974 to 1980 (Merritt, 1997). Note that as the storage time increased the RE

Table 2-2. Data from Hialeah ASR field trial

Cycle	1	2	3
Volume In (Mgallons)	41.9	85.0	208.0
Storage Time (days)	2	54	181
Volume out (Mgallons)	13.8	40.7	80.1
RE (%)	32.9	47.8	38.5
Q _{in} (gpm)	200 to 700	200 to 700	200 to 700
Q _{out} (gpm)	200 to 700	200 to 700	200 to 700

percentage decreased.

As can be seen on the table, the recharge and recovery rate varied during the test from 200 to 700 gallons per minute (gpm). The water quality data collected during the trial consisted of a limited set of parameters and did not include commonly monitored parameters such as dissolved oxygen, metals, and TTHMs. However, the water quality data that were collected were obtained at a high frequency. The source water was groundwater from the Biscayne Aquifer (Klein et al. 1975) that is of excellent quality. The chloride concentrations ranged between 55 and 65 mg/l, the TSS was low (less than 1.5 mg/l), and the color was moderate (20 to 30 color units). The ambient groundwater quality was brackish and poor. The chloride concentration was 1,200 mg/l, the TDS 2,700 mg/l, and the sulfate also high (Merritt, 1997).

During ASR recovery operations, the recovered water generally was a mix between the recharge and ambient water qualities. Some ferric hydroxide precipitates were noted periodically during recovery (Reese, 2002). No other geochemical reactions were reported. During the early portion of the first two recovery cycles, the recorded biochemical oxygen demand (BOD) of the recovered water was moderately high (0.5 to 1.3 mg/l) as compared to the BOD of the recharge water (0.2 mg/l). This could be an indication of biological growth around the ASR wellhead. Bacterial counts recorded during recovery show a pronounced increase in number and diversity by cycle 3 (Merritt, 1997). Chloride concentrations of the recovered water improved over each cycle and are shown on Figure 2-13, along with trend lines and regression equations.

Well clogging at the project site was moderate and was alleviated by periodic back flushing events lasting two to three hours duration (Merritt, 1997). No economic evaluations were completed for this test project but it is obvious that due to the low

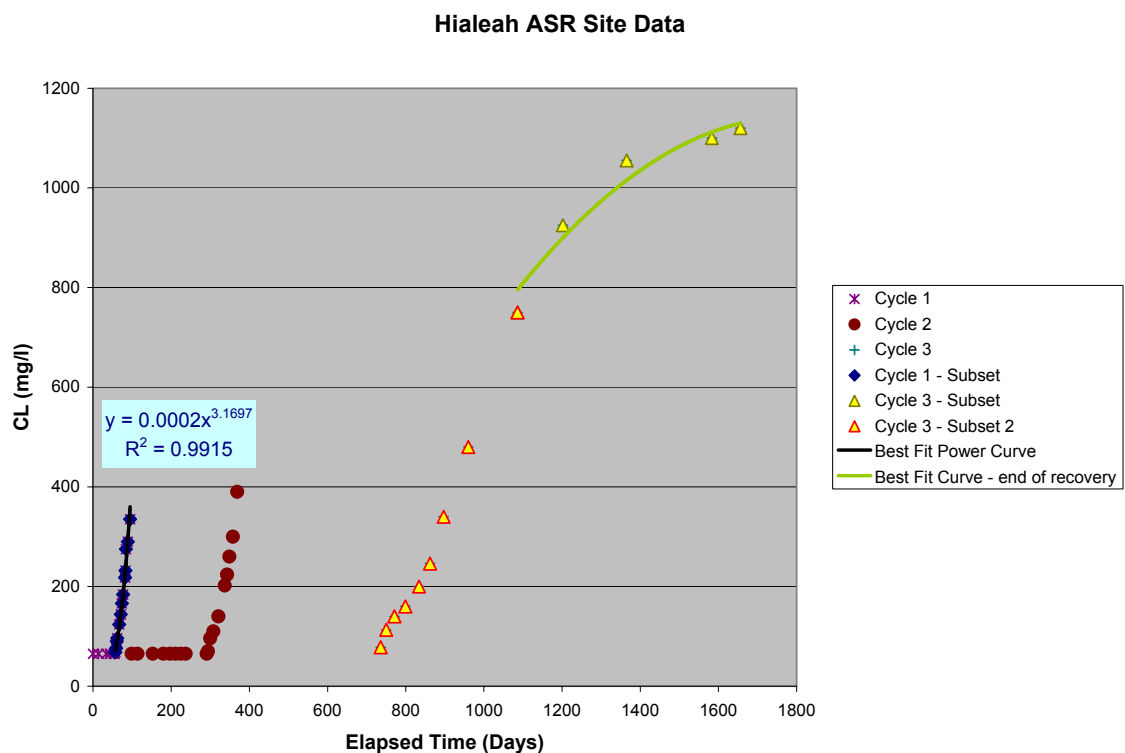


Figure 2-13. Hialeah cycle 1 through 3 recharge and recovery cycles, chloride water quality versus elapsed time in days

recovery efficiency of the project initially, the unit cost would not be as inexpensive as other sites that have operated longer.

Several lessons learned can be drawn from the experience gained at the Hialeah site. First, freshwater can be stored successfully in a moderately permeable limestone storage zone. Second, the performance of the ASR system can be reduced if storage times are long due to natural pre-existing gradients in the aquifer and possible buoyancy stratification of the freshwater due to density differentials. Third, well clogging can

occur in moderately permeable limestones but should be reversible through well back flushing or full re-development.

Tampa ASR site. The Tampa ASR project is located in the City of Tampa, Florida within the Rome Avenue Park (Peer Consultants, 1997). The project provides seasonal storage of excess freshwater from the Hillsborough River Reservoir for use during the dry season. A feasibility study completed in 1994 had recommended the Rome Avenue Park site as one of two optimal ASR locations (CH₂M Hill, 1994). The Rome Avenue Park site was selected based upon the following criteria (CH₂M Hill, 1994):

- Evaluation of the area's hydrogeology
- Water Quality
- Surrounding water use
- Contamination potential
- Facilities available to provide recharge water and use recovered water
- Site constraints
- Environmental issues
- Permitting and economic considerations

An ASR pilot well and associated monitoring wells were installed at the site in 1995 and pilot testing began soon after. The ASR well is completed in the Suwannee Limestone to 385 feet below land surface. The Suwannee Limestone is contained within the confined Upper Florida Aquifer System (Bush and Johnson, 1988). Figure 2-14 shows the general hydrogeology at the Tampa ASR site.

Aquifer performance testing and numerical modeling estimated the aquifer transmissivity to be 20,321 ft²/day with a storage coefficient of 9.8×10^{-5} (CH₂M Hill, 1998). In addition, numerical modeling utilized a longitudinal dispersivity of 20 feet and

an aquifer porosity of 15%. Specific capacity test data indicated moderate flow zones throughout the tested interval with the region between 300 to 320 feet below land surface characterized as highly productive (Peer Consultants, 1997).

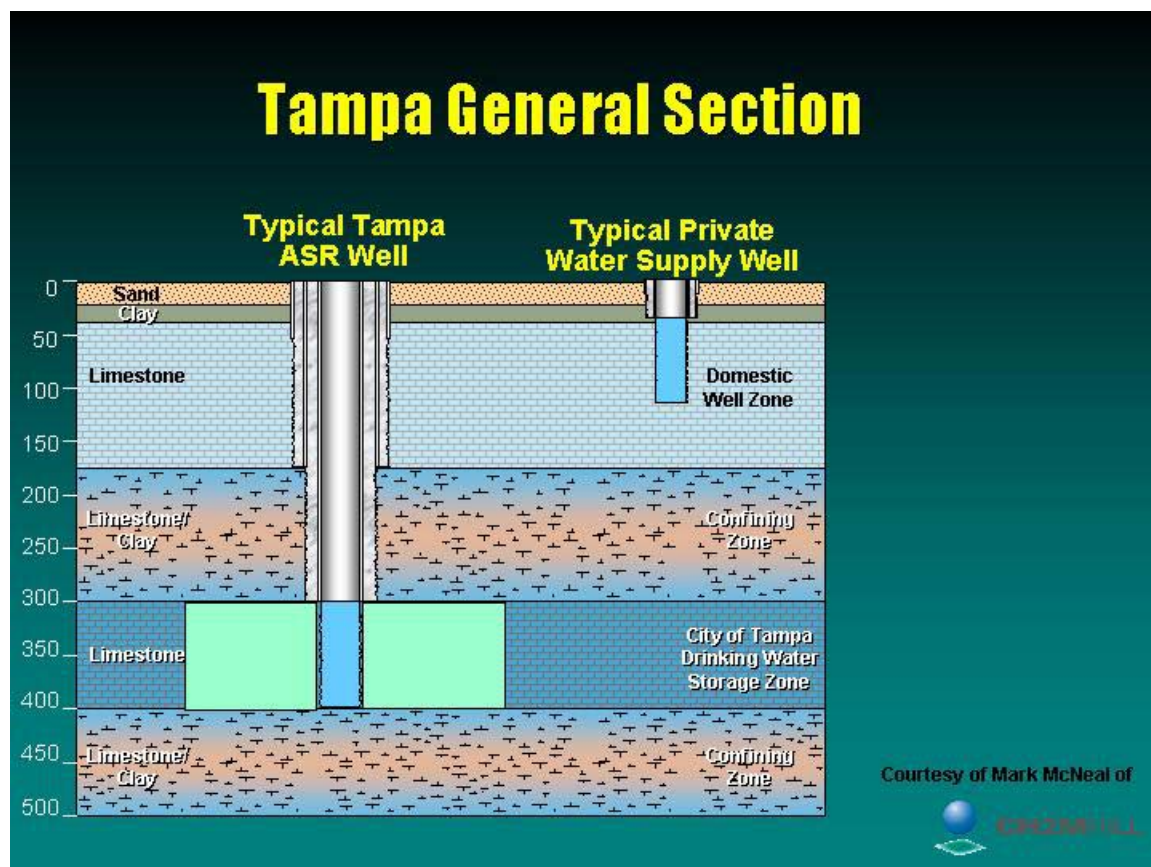


Figure 2-14. Tampa Rome Avenue Park ASR generalized site hydrogeology and location of selected ASR storage zone (elevations are feet below land surface; figure used with permission of Mark McNeal, CH₂M Hill)

The source water quality is treated drinking water of fair to good quality.

Concentrations of TDS vary between 270 to 370 mg/l, sulfate values vary from 90 to 150 mg/l, and turbidity varies from 0.08 to 0.36 NTU.⁵ The source water also contained low iron (less than 0.04 mg/l), moderate chlorine residual (1.5 to 2.0 mg/l), and moderate TTHMs (40 to 90 ug/l). The ambient groundwater at the site was slightly brackish with

⁵ TSS values were not recorded during pilot project cycle testing

TDS values ranging from 250 to 500 mg/l (Peer Consultants, 1997). During ASR recovery, mixing of the recharged water and the ambient groundwater was the dominant process, however, geochemical reactions were also indicated based upon the field data. Recovered water quality during ASR pilot cycles one and two revealed elevated levels of iron, calcium, ammonia, and radionuclides. The recovered water concentrations of these metals were higher than those recorded in either the recharge or ambient groundwater, indicating water-rock reaction (Arthur et al. 2002). Although arsenic was not included in the pilot testing program, current data suggest that arsenic is also released during ASR storage at the site (Arthur et al. 2002). The recovered water data also revealed high levels of turbidity recorded at the beginning of ASR recovery. Turbidity data were seven to ten times higher during recovery start than turbidity measured in the recharge water (Peer consultants, 1997). The turbidity value drops quickly and is close to background after one day. TTHM values drop quickly during ASR recovery so that at the end of cycle two recovery operations, the TTHM value is less than 1 ug/l. Lastly, at the end of cycle two recovery operations, the chloride concentration increases rapidly beyond the background level in the recharge or ambient water quality. This data indicates some brackish water upconing occurred during testing (Peer consultants, 1997). Motz (1992) discusses salt-water upconing in considerable detail. Figure 2-15 provides a schematic of upconing of more brackish water at the Tampa ASR project site. With the exception of the minor upconing and the geochemical issues, the ASR recovery efficiency was close to 100% during both test cycles.

Only minor well clogging issues have been observed to date. The single well ASR system has now been expanded to eight wells and only well ASR-5 continues to

experience moderate clogging that must be alleviated through well re-development (McNeal, 2004).

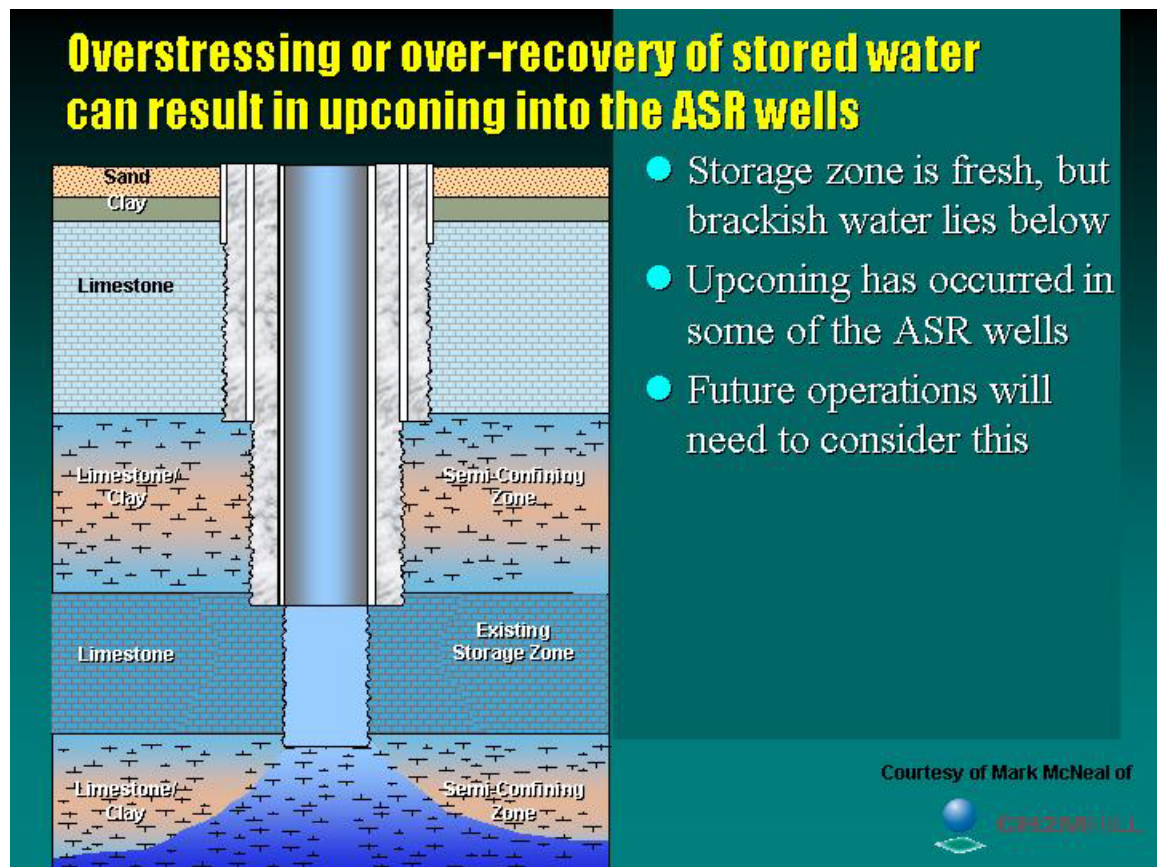


Figure 2-15. Tampa Rome Avenue Park ASR generalized brackish water upconing (deep aquifers within the FAS contain much higher TDS; figure used with permission of Mark McNeal, CH₂M Hill)

An economic evaluation of the system completed by the Tampa Water Department has estimated the unit cost of the project to be moderate at \$0.40 per thousand gallons recovered (McNeal, 2004). At first glance the cost estimate appears to indicate a very cost effective project, however, the cost provided does not include any costs for removing arsenic from the recovered water. The arsenic problems have led the Tampa Water Department to re-treat the recovered water to remove the arsenic prior to sending the water to the distribution system (McNeal, 2004). Without the arsenic problem, the cost

would be moderate largely due to the large amount of water that can be stored at the site for later recovery.

Multiple lessons learned are available from the Tampa ASR project. First, geochemical reactions can significantly affect overall project feasibility. Second, ASR storage within brackish water aquifers may have to consider the possibility of upconing of more highly brackish water from deeper aquifers. Lastly, well clogging can be problematic in permeable limestone aquifers, similar to unconsolidated aquifers, however, well back flushing usually can alleviate the problem more readily in carbonate units.

Peace River ASR site. The Peace River ASR project is located in southwest Florida within the Peace River watershed within Desoto County (CH₂M Hill, 1985). The Peace River ASR project is the oldest ASR project in Florida with initial investigations commencing in 1983 and full-time operations beginning in 1988 (Lehman and Waller, 1996). The ASR system is designed to provide long-term and seasonal water supply in conjunction with the Peace River (Lehman and Waller, 1996). The facility was originally envisioned to be able to supply up to six months of water supply that meets drinking water requirements during a long-term drought. The six-month time period was based upon a period of record analysis of the Peace River in 1985 (CH₂M Hill, 1985). In addition, besides the low river flows recorded during long-term droughts, the Peace River also experiences extensive algae blooms that result in difficult water treatment and taste and odor problems (CH₂M Hill, 1985). Over the last 20 years, the facility has been expanded several times and now consists of a large system composed of 21 ASR wells with 20 wells completed in the Suwannee Limestone Formation, one well completed in

the Tampa Limestone Formation, and one test well completed in the Avon Park Formation (Morris, 2004). During the well site selection exercise completed during initial investigations, a complete well inventory was completed of all local well users. The inventory was utilized in order to aid with selection of an ASR storage zone (CH₂M Hill, 1985).

The ASR wells onsite are screened in three different aquifers as noted above. Twenty of the wells are screened in the Suwannee Limestone of Oligocene Age. This aquifer is located within the Upper Floridan Aquifer System and is characterized as sandy, fossiliferous limestone and is located approximately 570 to 920 feet below land surface. Aquifer performance tests have been completed at multiple ASR wells. One well, S-5, produced objectionable quantities of sand during development and testing operations (CH₂M Hill, 1985). The average transmissivity of the confined Suwannee zone is 5,000 to 30,000 ft²/day with an average storage coefficient of 2.0×10^{-4} (CH₂M Hill, 1985). The Tampa Limestone zone transmissivity was estimated to be approximately 4,000 ft²/day with a storage coefficient of 8.0×10^{-5} and the deeper Avon Park may have a transmissivity of 150,000 ft²/day and a storage coefficient of 1×10^{-3} (CH₂M Hill, 1985).

The source water quality of the treated Peace River water is variable depending upon flow characteristics in the river. The TDS value can range from 160 to 596 mg/l (CH₂M Hill, 2003b). The chlorides range from 30 to 162 mg/l and the sulfate ranges from 32 to 175 mg/l (CH₂M Hill, 1985). The source water also contains moderate levels of metals such as calcium and TTHMs (levels ranged from 60 to 113 ug/l in the source water). The ambient groundwater quality within the Suwannee storage zone is slightly

brackish with TDS values ranging from 650 to 800 mg/l, chloride ranging from 151 to 206, sulfate ranging from 216 to 232 mg/l, and low levels of metals including arsenic (7 ug/l) and calcium (75 mg/l), (CH₂M Hill, 1985). Cycle testing originally began in 1984 using wells completed in the Suwannee Limestone. Table 2-3 summarizes data from the first five ASR recharge and recovery cycles. The RE percentage is defined by a TDS regulatory standard of 500 mg/l versus other brackish water sites that use chloride as a regulatory standard.

Table 2-3. Data from initial cycle testing at Peace River ASR site

Cycle	1	2	3	4	5
Volume In (Mgallons)	3.83	6.38	6.06	6.62	9.78
Storage Time (days)	3	1	1	1	17
Volume out (Mgallons)	12.29	6.75	6.98	5.90	9.58
RE (%)	321	106	115	89	98
Q _{in} (gpm)	200 to 700	200 to 700	200 to 700	200 to 700	200 to 700
Q _{out} (gpm)	200 to 700	200 to 700	200 to 700	200 to 700	200 to 700

Generally, the recovered water quality data from the initial cycles indicates mixing between the recharge and ambient water quality is the dominant geochemical process, however, early radionuclide data may reveal some geochemical water-rock reactions occurring during ASR storage. Gross alpha analyses at the beginning and end of each cycle test were tabulated during cycle testing. The end value measured for cycle 2 (21.4 pCi/L) indicates a gross alpha value nearly four times higher than the natural background reading of 5.8 pCi/L. A few other metals also exhibit possible water-rock reactions including iron, which was detected in one recovered water sample at a concentration of 2.35 mg/l, almost ten times higher than background measurements. TTHM concentrations in the recovered water generally declined during recovery operations as

the percentage of ambient groundwater in the recovered mixture increased. Typical TTHM concentrations measured at the end of recovery operations ranged from 18 to 28 ug/l (CH₂M Hill, 1985). Although TSS measurements were not available, turbidity values were initially high during the early portions of cycle test recovery and then receded quickly.

Most heavy metal data were not recorded during the early cycle recovery tests. More recent recovered water data have revealed more serious water-rock reactions that are suspected of releasing arsenic during ASR storage. Cycle testing of a new “west well field” at the Peace River complex has revealed the arsenic issue. Figure 2-16 depicts arsenic results from well # S11 during cycle 2 recovery. Due to the arsenic contained within the recovered water, the recovered water is re-treated in the onsite water treatment plant at a significantly higher cost. In addition, the arsenic issues have delayed a proposed ASR project expansion at the site (Morris, 2004). In reviewing the arsenic issue, acidization treatments using acid or carbon dioxide may be exacerbating the situation by exposing more of the carbonate section and by releasing additional oxygen into the aquifer.

Well clogging has not been a major problem at the Peace River project although small decreases in specific capacity have been noted (CH₂M Hill, 2003b). In general, specific capacity reductions are restored during ASR recovery operations.

An economic evaluation of the system completed by CH₂M Hill in 1985 has estimated the unit cost of the project to be moderate at \$1.00 per thousand gallons recovered (CH₂M Hill, 1985). An independent cost evaluation completed by the author for this report has estimated a current unit cost of \$0.72 per thousand gallons recovered

including re-treatment costs due to arsenic issues. Without the arsenic problem, the project would be very cost effective largely due to the large amount of water that can be stored at the site for later recovery.

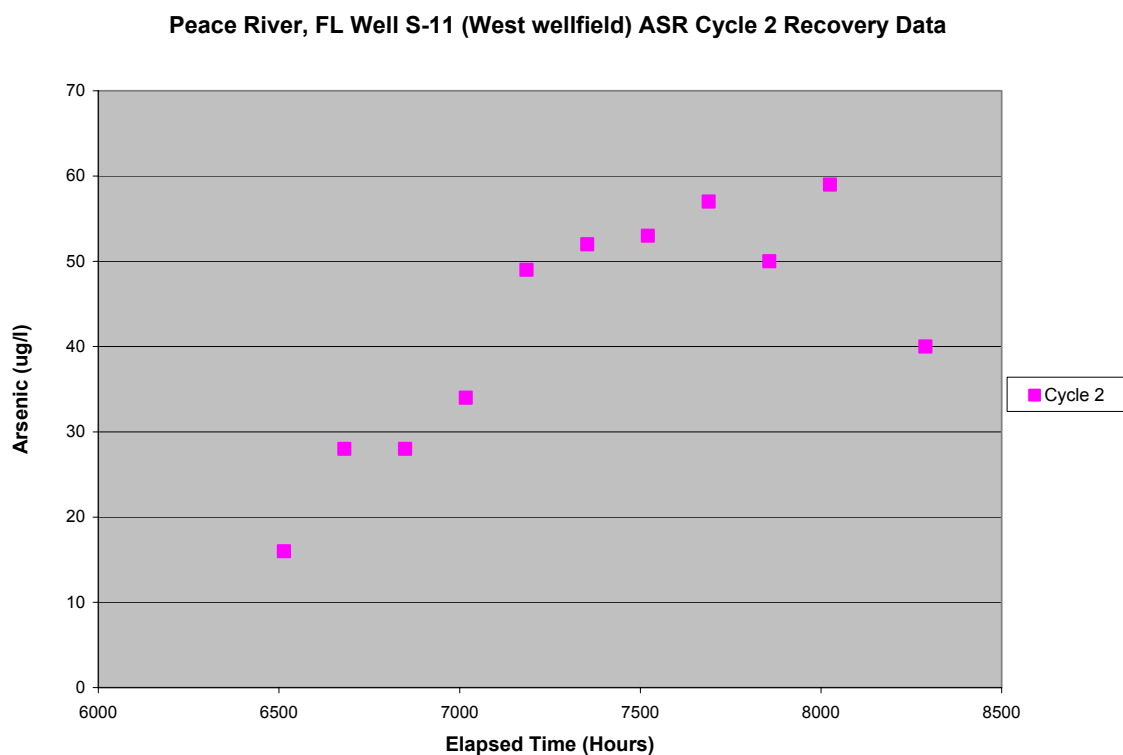


Figure 2-16. Cycle 2 recovery arsenic data for Peace River well # S11

Other Florida ASR sites. ASR has been collected and reviewed for 14 additional project sites in Florida. Many of these sites are similar in site hydrogeology, operational history, and use as potable water supply facilities. In addition, other authors have previously reported upon the attributes of a majority of these sites (Reese, 2002; Mirecki, 2004). Due to these facts, the remaining 14 Florida ASR sites will be discussed collectively. All of the sites with the exception of Cocoa Beach are located in south Florida. The Cocoa Beach site is located east of Orlando, Florida and west of the Atlantic Ocean. Several ASR test wells have been evaluated at Cocoa and reported upon previously (Merritt et al., 1983; Pyne, 1995). All of the remaining Florida sites are

utilized primarily for potable water supply as an aid to meet peak demands or for emergency water supply purposes.

All of the remaining 14 sites have ASR wells completed in the Lower Hawthorn Aquifer or within the Floridan Aquifer System. Table 2-4 provides a summary of hydrogeologic parameters for each site along with notes or references depicting source of the information. Data from a few of these sites were re-analyzed for this report and are noted as such. For example, the recovery efficiency estimated for the first cycle test at each site may have been re-analyzed as part of this report utilizing regression equations or general trend lines. Additional figures below depict field data from some of the 14 sites. Also, time versus concentration plots of cycle one recovery test data were plotted and displayed for this report. Lastly, long-term cumulative recovery efficiency was calculated based upon the available data and literature information.

Of the 14 sites evaluated, Cocoa Beach has been the most successful. Two separate Cocoa Beach test sites are included in the Table 2-4 below. The project has operated since 1987 and has undergone a series of expansions that provide the facility with over 10 MGD of ASR recovery capacity (Pyne, 1995). The source water is pumped from the Claude H. Dyal water plant and is generally of superior quality, although, it can be somewhat variable throughout the year. The ambient groundwater of the ASR storage zone is brackish and contains water with TDS concentrations of 900 to 2000 mg/l. The ambient groundwater also contains moderate concentrations of sulfate, calcium, and magnesium. Recovery efficiencies of the system routinely approach 100% with the exception of well # R2 where karst-like conduits may exist that effectively reduce the RE percentage due to mixing and dispersion. No major well clogging has been reported.

Table 2-4. Summary hydrogeologic data for 14 Florida brackish water ASR sites (NA means data not available)

Site Name	Transmissivity (ft ² /day)	Storage Coefficient	Cycle One RE %	# of Cycles reviewed for this report	Reference or Notes
Eastern Hillsboro	23,000	1×10^{-4}	10.30	1	Uram, 2005
Broward County WTP 2A	28,900	1.1×10^{-4}	14.30	3	Reese, 2002
Lee County Olga	9,000	5×10^{-5}	27.00	3	1) Reese, 2002 2) Mirecki, 2002 3) Re-analyzed RE% data
Sunrise Springtree	5,700	NA	23.20	6	1) Reese, 2002 2) Re-analyzed RE% data
Delray Beach	8,000	NA	19.00	7	1) Mirecki, 2004 2) Re-analyzed RE% data
Fiveash	19,500	NA	7.50	6	1) Reese, 2002 2) Mirecki, 2004 3) Re-analyzed RE% data
Manatee Rd	10,000	1×10^{-4}	5.00	4	1) Reese, 2002
Marco Lake	12,000	6.5×10^{-5}	23.00	5	1) Reese, 2002 2) Mirecki, 2004 3) Re-analyzed RE% data
Miami-Dade WWF	15,400	4.25×10^{-4}	8 to 16.00	3	1) Reese, 2002 2) Mirecki, 2004 3) Re-analyzed RE% data
Cocoa Beach 1	9,024	2.65×10^{-4}	95.00	10	1) Pyne, 1995
Cocoa Beach 2	210,000	NA	33.90	5	1) Merritt et al., 1983
Fort Myers Winkler ave	28,000	NA	1.58	2	1) Mirecki, 2004
St. Lucie	6,000	2.10×10^{-4}	3.00	3	1) Reese, 2002
Shell Creek	1,300	NA	30.00	4	1) Reese, 2002
Lee County North	8,500	3.27×10^{-4}	9.80	3	1) Mirecki, 2004 2) Re-analyzed RE% data

Cocoa Beach has experienced some issues with brackish water upconing also (Pyne, 1995).

Several of the sites including Eastern Hillsboro, Miami-Dade WWF, and Delray Beach, have experimented with developing a “target storage volume”. The target storage volume (TSV) approach was originally proposed by Pyne (1995) and includes recharging a large volume of water to build a mixing buffer between the freshwater and the ambient groundwater. By investing a large amount of freshwater in the initial segments of an ASR program, recovery of some programmed volume of freshwater can be assured. For example, the Miami-Dade WWF site used the TSD approach during cycle one and recharged approximately 360,000,000 gallons of freshwater. If the operational need is 50,000,000 gallons during peak summer season, the actual RE percentage need only be greater than 13.9% in order to meet the need. At the Eastern Hillsboro site, cycle one recharge included injection of approximately 497,000,000 gallons of freshwater and recovery of almost 51,000,000 before the chloride concentration of the recovered water exceeded 250 mg/l. The Delray Beach site recharged approximately 213,000,000 gallons of freshwater and recovered almost 50,000,000 before regulatory limits were reached. Figure 2-17 displays the cycle one recharge and recovery data along with regression equations for the best-fit recovery curve. Due to the brackish nature of the ambient groundwater, the recovery curve is quite steep during cycle one, with the best-fit equation a power curve.

Several of the sites have experienced geochemical issues with arsenic similar to the Tampa and Peace River sites discussed previously. Both Lee County North and the Lee County Olga ASR sites have observed arsenic in the recovered water. Mirecki (2004)

has discussed geochemical issues at both sites along with other sites located in southwestern Florida. In addition to the arsenic contained within the recovered water, the two Lee County sites have behaved differently during ASR recovery. Although the two

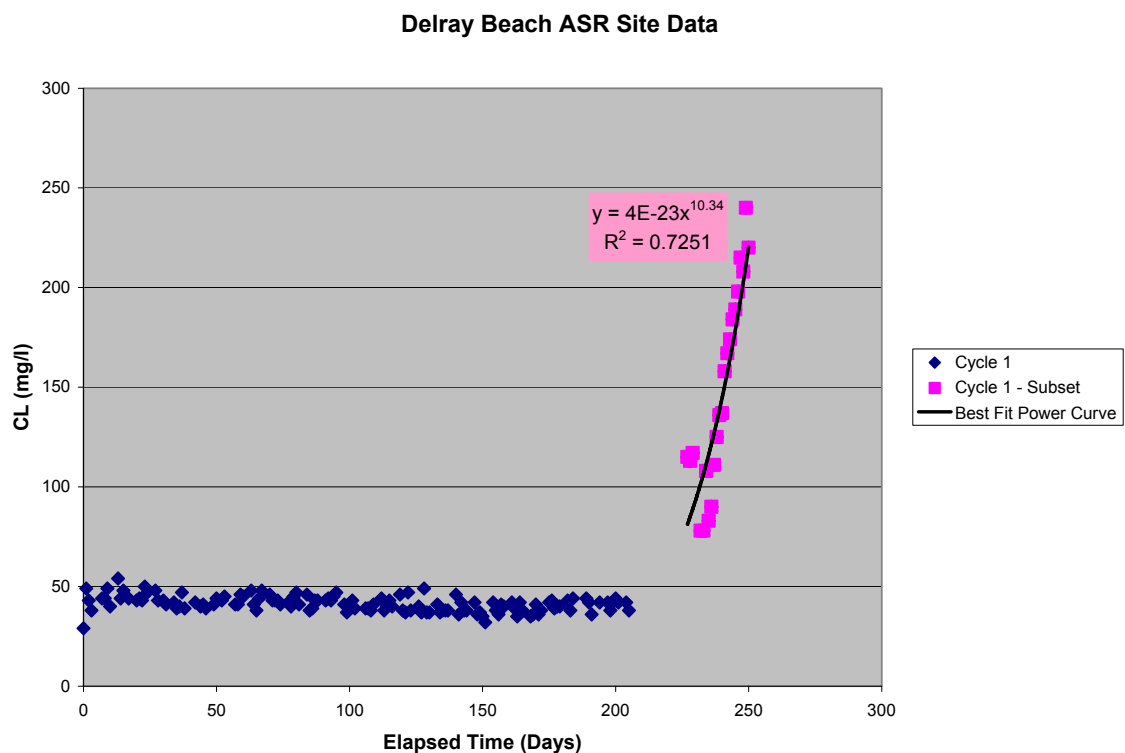


Figure 2-17. Cycle 1 recovery data for Delray Beach ASR well

sites have similar hydrogeology and similar ambient groundwater quality, the RE percentage for each during cycle one was quite different. Recovery curves for both sites are shown here as Figures 2-18 and 2-19.

The Lee County North site exhibits a very steep recovery curve even after one full cycle. The RE percentage of cycle one was 9.8% while cycle two was not improved much with a RE percentage of 11.70%. One reason for this apparent lack of improvement may be the long storage duration employed during cycle two. A storage duration of approximately 168 days was utilized before recovery operations could commence. During this time, the data indicate that buoyancy stratification in the aquifer

did indeed occur as depicted by a modest rise in chloride concentration from the end of recharge to the start of recovery. The Lee County Olga site also exhibits a steep recovery

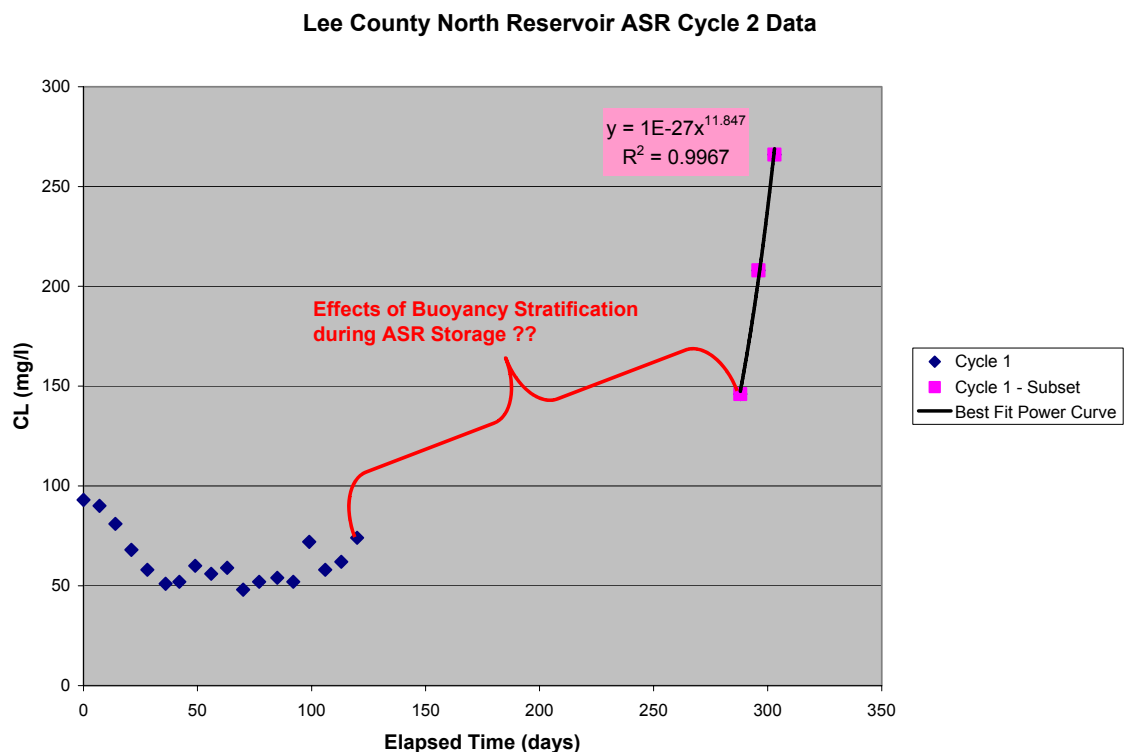


Figure 2-18. Cycle 2 recovery data for Lee County North ASR well

curve, similar to Lee County North, however, no obvious buoyancy stratification effects are evident from the cycle one recharge and recovery data. This data is shown on Figure 2-19.

Steep recovery curves are also evident for the Broward County 2A, Fiveash, and Fort Myers Winkler Avenue sites. These recovery curves are shown on Figures 2-20 to 2-22. All three sites had minimal storage periods during the cycle one recharge and recovery cycle. Broward County and Fiveash also exhibit a fairly consistent recharge influent chloride water quality. That is not the case with the Fort Myers Winkler Avenue site where the chloride concentration of the recharge varies between 150 and 190 mg/l. Using the best-fit power recovery curves for each site enabled the author of this report to

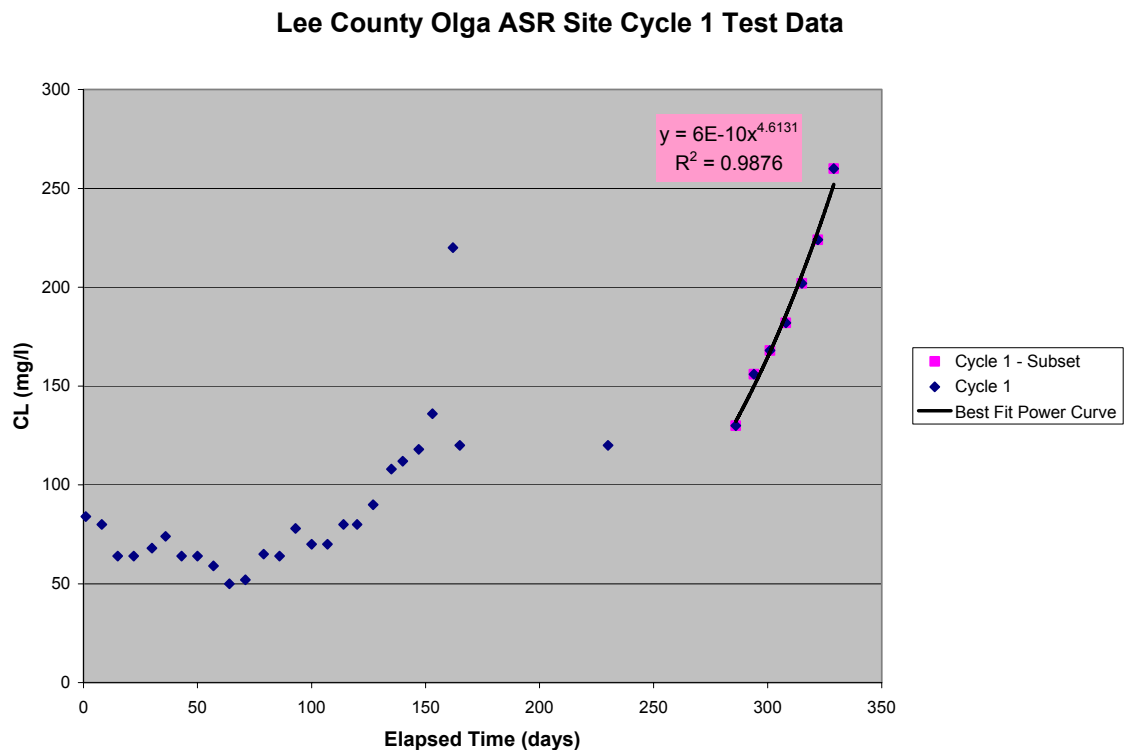


Figure 2-19. Cycle 1 recovery data for Lee County Olga ASR well

recalculate the RE percentage for each site. All three of sites had low RE percentages for the initial project cycles with values of 9%, 7.5%, and 1.58%, respectively. Besides the initial cycle testing recovery curves, each of the sites was evaluated to determine the cumulative recovery efficiency or CRE, as was developed by Kimbler et al. (1975).

Plotting the CRE versus the cycle number is an excellent way of observing ASR performance over the long term. As was noted previously, as the cycle number increases, so does the individual cycle RE percentage. The same is true of the CRE percentage.

Figure 2-23 depicts the CRE for a number of the 14 remaining Florida ASR sites. In general, the CRE increases with subsequent cycles. Using the TSV approach seems to improve the CRE more rapidly than conventional ASR approaches by allowing the creation of the mixing buffer. Delray Beach and Miami-Dade WWF have the highest CRE of the sites analyzed on the figure. These sites utilized the TSV approach in their

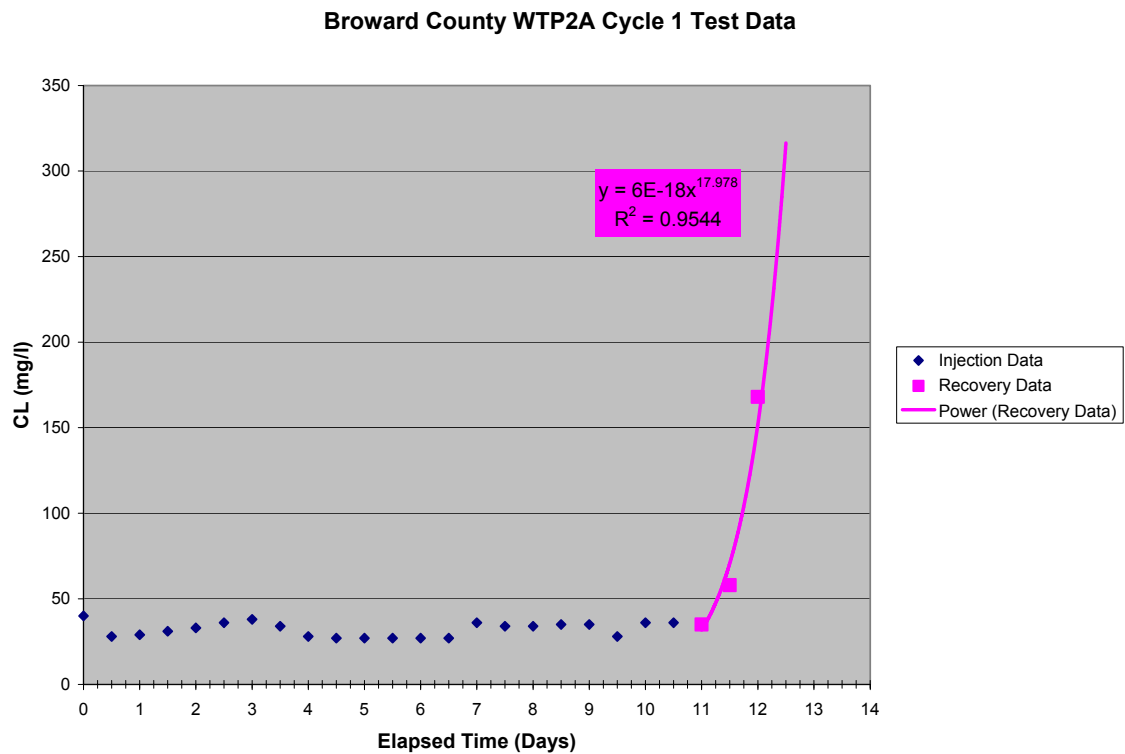


Figure 2-20. Cycle 1 recovery data for Broward County WTP ASR well

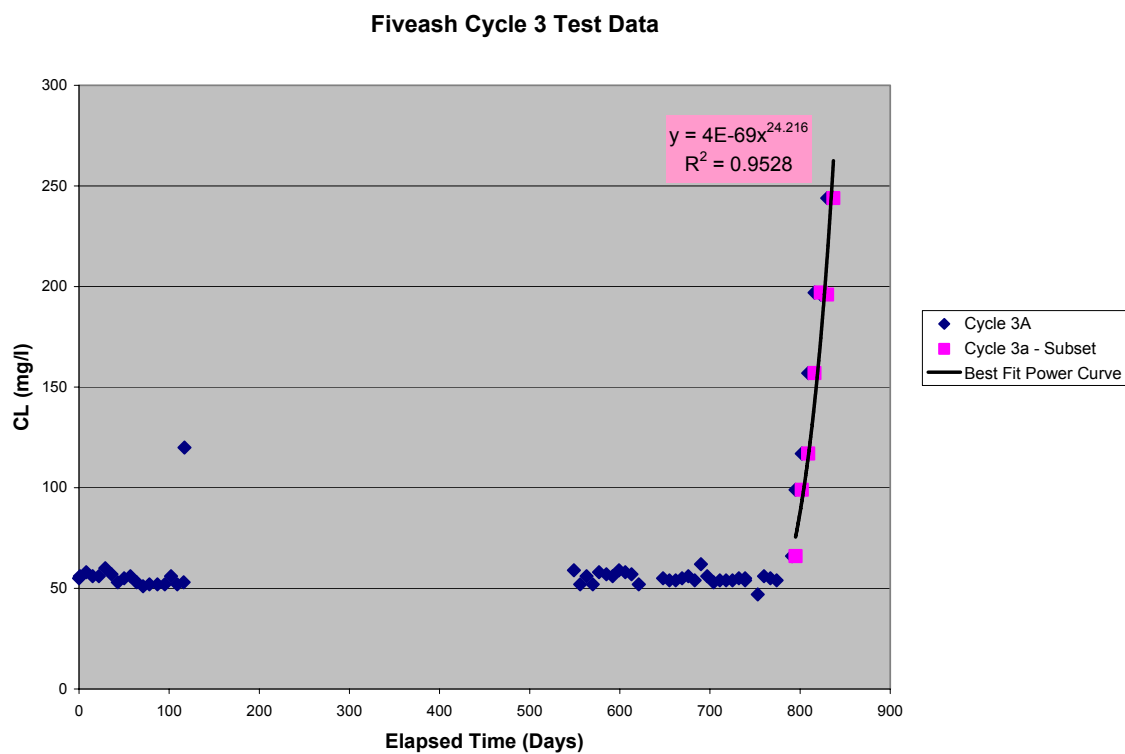


Figure 2-21. Cycle 3 recovery data for Fiveash ASR well

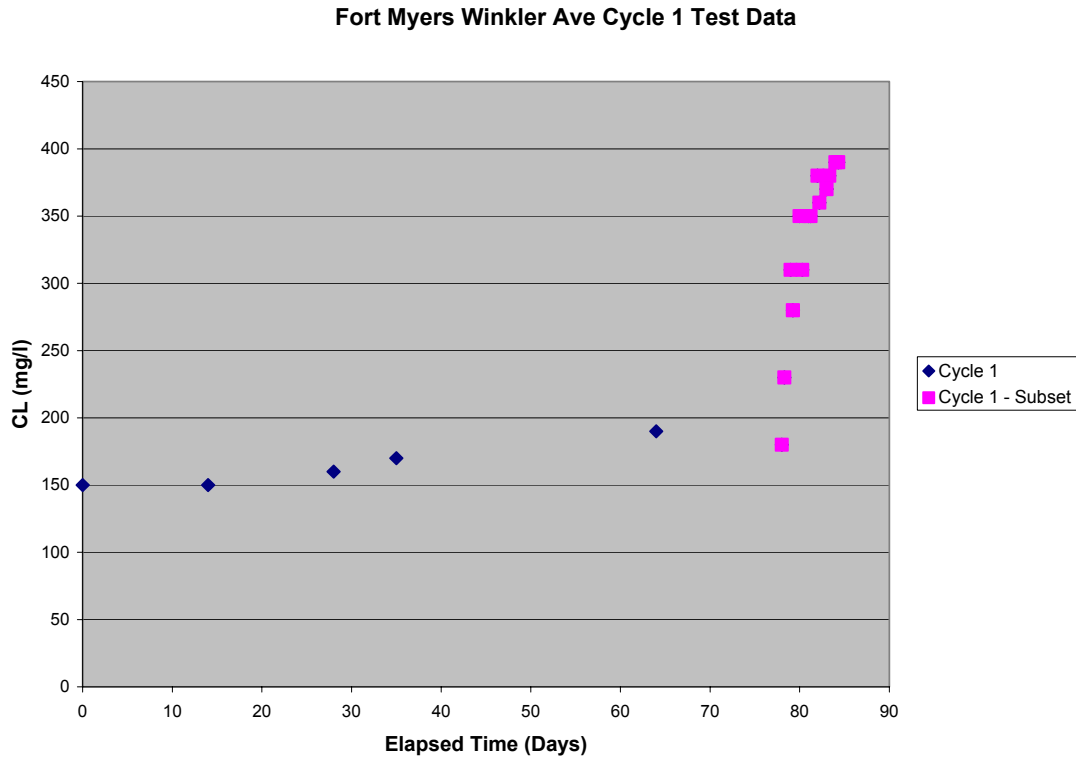


Figure 2-22. Cycle 1 recovery data for Fort Myers Winkler Avenue ASR well operations. Noteworthy CRE decreases are also manifest on Figure 2-23. Hialeah, Sunrise Springtree, and Fiveash all had at least one cycle where the storage duration between recharge and recovery was long. In all three cases, the long storage time results in a marked decrease in CRE. The cause of the decrease is probably due to either buoyancy stratification effects or the effect of a pre-existing aquifer gradient. Obviously, if the storage duration is long enough, both phenomena could occur and substantially reduce the ASR performance for the cycle affected. Another operational issue that can affect the CRE over time is recovery of a large volume of freshwater during a cycle. This action reduces the mixing buffer between the freshwater and the more brackish ambient groundwater. This effect is evident at the final cycle plotted for Delray Beach where the CRE drops by 4% rather than continuing to increase.

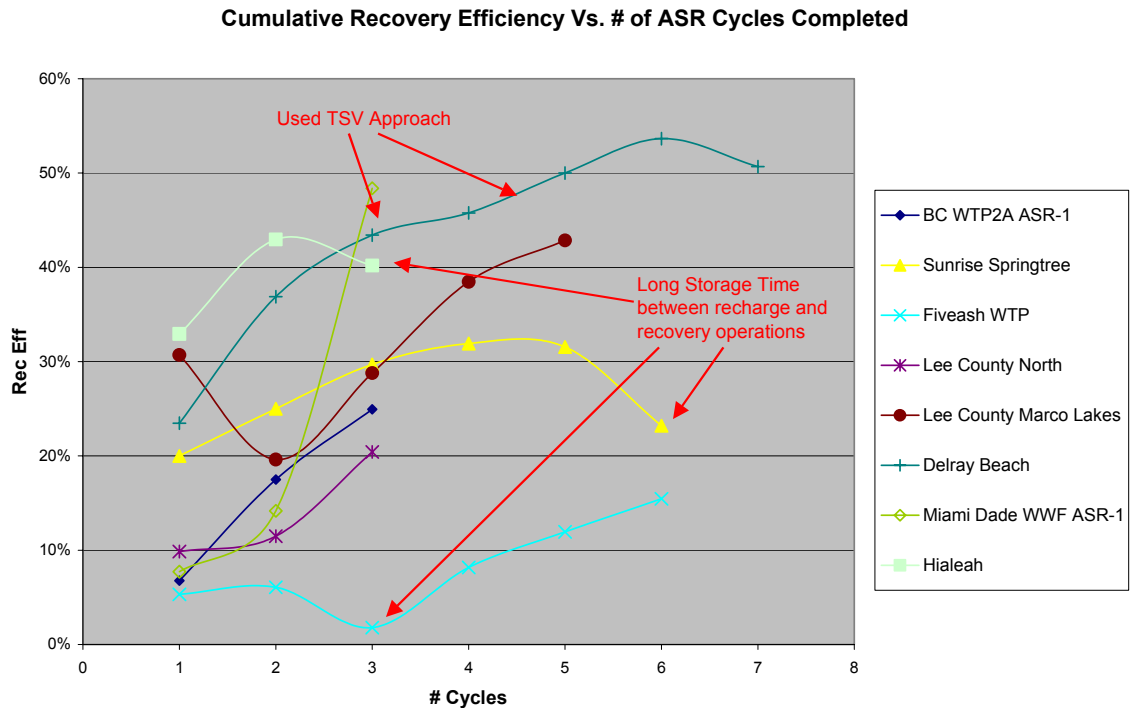


Figure 2-23. Cumulative Recovery Efficiency (CRE) versus cycle number for eight brackish water Florida ASR sites

International brackish water sites. All five of the remaining brackish water ASR projects are located outside of the United States. Four of the sites are located in Australia including Andrews Farm, Clayton, Willunga, and Bolivar, while one site is located in India (Northern India). All of these sites have been discussed in the literature and are only briefly discussed in this report. In general, four of the sites are utilized in support of agriculture through the provision of supplemental irrigation water, while one site (Clayton) provides a town water supply when surface lake water is unavailable due to algae contamination (Gerges et al. 2002b).

The hydrogeology of the five sites is diverse and includes both confined and unconfined aquifer project sites. Three of the sites have ASR wells completed in confined carbonate, sandy limestone aquifers including Andrews Farm, Willunga, and Bolivar (Barry et al. 2002; Sibenaler et al. 2002; Vanderzalm et al. 2002). The remaining

two sites (Clayton and Northern India) have ASR wells completed in unconfined aquifers containing unconsolidated sediments (Gerges et al. 2002b; Malik et al. 2002a). The Clayton ASR well is located in an indurated and highly permeable, karstic aeolianite aquifer (Barnett et al. 2000). Key hydrogeologic information along with cycle testing information and relevant references utilized in the summary is provided in Table 2-5. For the international sites, RE percentage has been calculated using a TDS standard rather than a chloride standard more common in the United States. Therefore, the data for Willunga, Bolivar, and Clayton cannot be compared directly to the Florida ASR sites.

Table 2-5. Summary hydrogeologic data for five international brackish water ASR sites (NA means data not available)

Site Name	Transmissivity (ft ² /day)	Storage Coefficient	Cycle One RE % (TDS used instead of Cl)	# of Cycles reviewed for this report	Reference
Andrews Farm	2,200	2.8×10^{-4}	10.30	3	1) Barry et al., 2002 2) Barnett et al., 2000
Clayton	16,146	Unconfined Aquifer	5.0	1	1) Gerges et al., 2002b 2) Barnett et al., 2000
Willunga	970 to ?	NA	6 for cycle 3; 60 for cycle 4	3	1) Sibenaler et al., 2002 2) Buisine and Oemcke, 2002
Bolivar	1,615	5.0×10^{-4}	60.80	1	1) Vanderzalm et al., 2002 2) Pavelic et al., 2005 (submitted)
Northern India	1,614	Unconfined Aquifer	43.50 to 55	1	1) Malik et al., 2002a 2) Malik et al., 2002b

Available data from Andrews Farm and Northern India permitted an estimate to be developed for RE percentage based upon a chloride standard. These data are discussed further in later sections of this report.

The Andrews Farm ASR site is located on the northern margin of metropolitan Adelaide in Australia (Barnett et al. 2000). The project provides water supply for irrigation purposes. An experimental ASR pilot project was conducted at the site from 1993 to 1998 (Barry et al. 2002). The ASR storage zone is a confined carbonaceous sand known as the “T2” aquifer (Barry et al., 2002). The hydrogeologic parameters as determined through aquifer testing are displayed in Table 2-5.

Source water for the project is derived from passively treated storm water from a housing development. The quality of the source water is fair to good with low TDS (250 mg/l), moderate nitrates, moderate TOC (7 mg/l), and high TSS (1 to 170 mg/l). The ambient groundwater contained poor quality water with TDS of 2,200 mg/l (Barnett et al. 2000). Five major recharge and recovery cycles were conducted as part of the test with approximately 66 million gallons of recharge injected into the T2 aquifer (Herczeg et al. 2003). In general, well clogging was only moderate and was easily reversed through periodic well redevelopment. Redevelopment usually amounted to a recovery of 50,000 to 500,000 gallons of water from the ASR well (Barry et al. 2002). Geochemical studies noted some minor carbonate dissolution and sulfide mineral oxidation during cycles one to four, but none were noted during cycle 5 (Herczeg et al. 2003). ASR RE percentage ranged from 10 to 30% for the first few cycles and then rose steadily to 60% in the fourth year of testing operations.

The Clayton Township ASR project is located south of Adelaide, Australia at the lower end of the Murray River system (Gerges et al. 2002). The Clayton project is unique in that it is one of the few ASR projects in the world storing water in an aquifer containing seawater. The ASR well serves to supplement the town water supply during summer months when the main water supply from Lake Alexandria is subject to occasional blooms of blue-green algae (Barnett et al. 2000). Blue-green algae are a type of cyanobacteria that produce toxins. The ASR project is designed to function as an emergency supply capable of supplying the town with up to 18 million gallons of water during the summer months if the lake water is unavailable due to algae blooms (Gerges et al. 2002). The ASR storage zone is an unconfined carbonate limestone of Tertiary Age. The relevant hydrogeologic parameters estimated through step-drawdown tests are displayed in Table 2-5 above.

The source water quality is fair to good quality with moderate TDS (430 mg/l), low chloride (158 mg/l), coliforms (5,600 per 100 ml), and high TSS of up to 50 mg/l (Gerges et al., 2002). The ambient groundwater is seawater with at TDS of approximately 30,100 mg/l, chloride of 17,050 mg/l, and high iron (0.44 mg/l). Typically a large sacrificial freshwater lens is created through high recharge rates. Buoyancy stratification then degrades the lens over time so that approximately 78 million gallons of recharged freshwater must be injected into the aquifer to allow for summer peak demand recovery to be met (Gerges et al. 2002). Although the ASR RE percentage is usually only five to ten percent, the recovered water has very high clarity allowing for the use of UV disinfection rather than chlorination (Barnett et al. 2000). Therefore, the ASR infrastructure and pumping costs are substantially cheaper than other alternative water

sources (Gerges et al. 2002). Well clogging has not been a problem but Gerges et al. (2002) noted that long-term clogging may become a problem so that monitoring is continuing. No geochemical issues have been reported. No economic evaluation of the project was contained within the literature reviewed for this report.

The Willunga ASR project is located adjacent to the Christies Beach Waste Water Treatment Plant in Willunga, Australia. The Willunga project was developed in order to test the feasibility of storing reclaimed wastewater in the carbonate Willunga Formation (Sibenaler et al. 2002). The project site was selected based upon a number of criteria including:

- Favorable aquifer thickness and transmissivity
- Ambient groundwater salinity
- Proximity of existing domestic users (e.g., no well users within 1.2 miles from site)
- Access to reclaimed water
- Infrastructure for disposal of recovered water
- Proximity to power and site security

The hydrogeologic parameters of the Willunga limestone are displayed in Table 2-5 above. It is important to note that the actual transmissivity may be higher than reported due to the presence of conduit flow zones in the aquifer. According to Sibenaler et al. (2002), the aquifer exhibits a dual-porosity character in the immediate vicinity of the ASR well whereas beyond a radius of 600 feet, the behavior is more typical of a porous media flow system. In addition, caliper logs revealed highly fissured areas within the initial exploration hole.

The source water quality from the wastewater plant was fair with high TDS (819 mg/l), moderate TSS (range from 1 to 31 mg/l with most values less than 1.0), and high phosphorus (8.77 mg/l). The ambient groundwater is brackish with TDS of 1,100 to 2,000 mg/l. Well clogging issues were minor and reversible during recovery operations.

Therefore, a planned well development strategy was not undertaken for the project based upon field data (Sibenaler et al. 2002). Monitoring well data collected during cycle testing operations revealed highly heterogeneous mixing patterns occur at the site due to the dual-porosity nature of the storage zone. The nature of the aquifer likely enhanced diffusive exchange with the ambient groundwater entrained within the less permeable portions of the aquifer (Sibenaler et al. 2002). The RE percentages during cycle testing improved from 6% during cycle 3 to 60% during cycle 4 although RE was defined using a regulatory limit of 2,000 uS/cm rather than TDS or chloride. Sibenaler et al. (2002) also notes that improvements in RE could be seen by locating recovery wells down gradient of the ASR injection well field, whereas, RE reductions could occur if the storage time between recharge and recovery is long.

Another interesting study completed during pilot testing was a pathogen inactivation study carried out using specially designed chambers designed to be placed within onsite monitoring wells (Sibenaler et al. 2002). The study results indicated that bacteria concentrations declined significantly ranging from 1.5 to 5 log cycle reduction in three weeks. Field trial evidence supports the conclusion that in-situ microorganisms were responsible for the high rates of bacteria degradation. No economic evaluation of the project was contained within the literature reviewed for this report.

The Bolivar ASR site is located adjacent to the Bolivar Waste Water Treatment Plant north of the City of Adelaide, Australia (Vanderzalm et al. 2002). The Bolivar ASR project is exploring the feasibility of storing reclaimed wastewater in winter months for subsequent recovery in dry summer months for irrigation purposes. The ASR storage zone is the confined, brackish T2 aquifer discussed previously in this report. The T2

aquifer is a sandy carbonate limestone aquifer of marine origin (Vanderzalm et al. 2002). The hydrogeologic parameters as estimated based upon field data and numerical modeling are shown in Table 2-5. Numerical modeling and field data also estimated an aquifer porosity of 45 to 47% and a nearfield longitudinal dispersivity of 6.5 to 16.5 feet (Pavelic et al. 2005). Evaluations also demonstrated that the dispersivity value increases with increasing distance from the ASR injection well (Pavelic et al. 2005).

The source water quality was fair with chloride concentrations ranging from 370 to 490 mg/l. The ambient groundwater quality was brackish with chloride concentrations ranging from 700 to 1,000 mg/l (Pavelic et al. 2005). Multiple recharge and recovery cycles were completed at the site with a total of 65 million gallons of reclaimed water recharged and approximately 40 million gallons of water recovered over the two year testing duration (Vanderzalm et al. 2002). During ASR storage, monitoring data revealed that aerobic oxidation of recharged organic matter and denitrification occurred near the recharge point while minor dissolution of the limestone matrix (mainly calcite dissolution) was also detected (Vanderzalm et al. 2002). The monitoring data also strongly suggests that natural bacterial communities flourished during ASR storage as seen in the rapid increase in the concentration of DOC at the ASR well. The increase in DOC is attributed to metabolism of organic matter and biomass/nutrient recycling (Vanderzalm et al. 2002). Additional data suggested that both limited sulfate reduction and methanogenesis occurred in the storage zone (Vanderzalm et al. 2002). Much of the field geochemical data was consistent with predictions based upon laboratory column experiments completed in 1999 (Rinck-Pfeiffer, 2000). The column experiments, using reclaimed water from the Bolivar plant, predicted problems with high biological activity

and calcite dissolution. Well clogging problems were noted to be minor during the testing program. No economic evaluation of the project was contained within the literature reviewed for this report.

The Northern India ASR project is located in Badyan Ranghran Hisar Haryana, India (Malik et al. 2002a). The project utilizes large diameter “tube wells” to recharge and recover excess water into unconsolidated sands and gravels. The aquifer is an anisotropic unconfined to semi-confined aquifer that is very permeable (Malik et al. 2002b). A series of recharge and recovery tests were completed to assess ASR storage of irrigation water within a brackish-water aquifer. Limited data are available from this test but the RE data are reported (Malik et al. 2002b). The RE is reported referenced to a TDS standard but chloride data presented allow development of an estimate of RE using a chloride drinking water standard instead. These data are discussed further in Chapter 3 of this report.

Several important observations can be made based upon the reported data. First, the RE improves from cycle to cycle due to enlargement of the freshwater buffer zone. Second, RE improves with increasing volume injected. Third, temperature differences between the recharge water and the ambient groundwater do affect the performance of an ASR well due to viscosity effects on the apparent transmissivity. Table 2-5 presented earlier in this report provides additional information from this project.

Comparison of Existing ASR Site Data

The 50 sites chosen for discussion in this report represent a multitude of different geologic environments as well as a wide variety of locations in the United States, England, Australia, Africa, and India. Some of the ASR projects were constructed for testing purposes only (Bureau of Reclamation, 1996a; Bureau of Reclamation, 1997),

while others were planned to for irrigation or municipal water supply (Bureau of Reclamation, 1994; Pyne, 1995; CH2M Hill, 1999; CH2M Hill, 2001; Portland Water Works, 2001; Sibenaler et al. 2002; Calleguas, 2004; Groundwater Solutions, 2004; Mirecki, 2004; Pavelic et al. 2005). The setting, background, and operational histories of twenty (20) non-brackish water ASR projects were reviewed along with thirty (30) brackish water ASR projects. After the compilation and review of the data was completed, the various sites were compared and contrasted to reveal key similarities and differences. Common lessons learned from site operating data were of special interest.

Non-brackish water ASR sites

The non-brackish water ASR sites were located in the United States, England, and Africa. Twenty representative sites were chosen where basic information was available and operational summaries existed or could be easily developed for this report.

The surveyed ASR projects are located in three countries and eleven states within the USA. The geology varies across the sites and includes both bedrock (9 sites) as well as unconsolidated sediments (11 sites). Of the bedrock sites, the predominant rock type is sedimentary (6 of 9 sites) with two located in an igneous rock (basalt) type and one within a metamorphic (quartzite) type. Two of the six sedimentary rock sites are composed of carbonate sequences including limestone or chalk, while the remaining two sedimentary rock sites are composed of predominately sandstone. For the case of the 11 sites situated in unconsolidated geologic environments, sands and gravels dominated the composition of the sediments. Of the 20 sites reviewed, 15 of them had the ASR wells completed within a confined aquifer while five sites utilized an unconfined aquifer as the ASR storage zone. Of the confined aquifer sites, the storage coefficient was estimated

for 10 sites. The storage coefficient for the ASR sites ranged from 4.0×10^{-6} to 3.0×10^{-3} with a geometric mean of 1.91×10^{-4} .

Aquifer transmissivity data were available for 19 out of the 20 sites reviewed. The aquifer transmissivities for the ASR sites ranged from 856 ft²/day to 38,000 ft²/day with a majority (12 of 19 sites where data were available) having values less than 10,000. The geometric mean transmissivity for the sites was calculated to be approximately 5,022 ft²/day. All of the sites underwent some form of pilot testing during an early stage of the project.

Water quality issues were important at half of the twenty sites reviewed. Disinfection by-products including TTHMs and HAAs were monitored at 17 of the 20 sites reviewed. Of the 17 sites studied, only two sites (Las Vegas and Lancaster) had reported major problems with DBP formation in-situ. The remaining sites reported DBP levels declined in ASR storage due to mixing, hydrodynamic dispersion, and biodegradation. Only three sites noted concentrations of DBPs that occasionally exceeded regulatory standards of the United States. This is not unexpected as the reported residual chlorine for four of the sites was relatively low ranging from 0.79 to 3.5 mg/l. A data review of several sites (Oak Creek, Beaverton, Hilton Head, Lancaster, and Las Vegas) strongly indicated biodegradation of DBPs. Generally for these sites, the level of HAAs was noted as declining more rapidly than associated TTHMs. A few studies reported that of the TTHM monitored or evaluated, chloroform was the most resistant to biodegradation. Several of the project sites have opted to utilize other forms of disinfection besides chlorination in order to reduce the generation of TTHMs in the source water for an ASR project. The Calleguas ASR project is noteworthy in this

respect as it employs chloramines for disinfection and has not had any regulatory issues. The Salt Lake City project utilizes UV disinfection while the Green Bay project utilized an onsite ozonation facility.

Well clogging issues were monitored at 18 of the 20 sites reviewed. Of these 18 sites, 10 reported little to no well clogging issues at all. Seven of the 18 sites reported moderate well clogging that required regularly scheduled well back flushing or redevelopment. Of the 18 sites reviewed, only one site (Portland) reported a major loss in well recharge or recovery efficiency. The cause of this loss is still under investigation (Moncaster, 2004). Suspended solids levels were reported at 17 of the 20 sites. Nine of the sites reported average TSS concentrations of less than 1.0 mg/l while seven of the sites reported averages of between 1.0 and 5.0 mg/l. The Beaverton site reported average influent TSS concentrations as high as 10 mg/l. Interestingly, five of the seven sites that reported moderate well clogging issues, also reported average TSS influent concentrations of greater than 1.0 mg/l. Beaverton did not report any major clogging problems although the TSS was the highest of the 20 sites reviewed. Moderate well clogging has been observed at the Hilton Head Island ASR site. Specific injectivity of the ASR well declined rapidly during cycle 1 recharge operations. Due to the carbonate nature of the storage zone and high aquifer transmissivity, this was somewhat unexpected. It is possible that in addition to the importance of minimizing influent suspended solids concentrations; the total mass loading of suspended solids may also be an important design consideration. Luckily, the capacity of the ASR well was fully restored through redevelopment and back-flushing operations. Similar problems were noted at the Salem, Oregon site, although that site also has a very high aquifer

transmissivity. ASR wells in the Portland, Oregon area have experienced major well clogging problems that have reduced the specific injectivity of the ASR wells by 5 to 37%. The Portland Water Works is currently investigating if the loss of specific capacity is permanent.

Although clogging by TSS was not reported as a major issue, the consultant at the Beaverton site reported that air entrainment caused by cascading water reduced the apparent transmissivity of the storage zone (Groundwater Solutions, 2004). Down hole control valves eliminated this issue as it has in other sites (Pyne, 1995). The Alamogordo project utilized an eductor or drop pipe to eliminate air entrainment issues. Air entrainment problems that reduced well capacities were also reported at the Huron project and Highlands Ranch. The loss of well efficiency due to viscosity differences between the colder recharge water and the warmer ambient groundwater was reported at the Denver Basin test project.

Geochemical issues were reported on, and evaluated at 15 of the 20 sites reviewed. Of the 15 sites where data were available, 13 of the sites reported none to minor geochemical issues. Generally, minor geochemical issues noted included low levels of calcite dissolution, minor enrichment of metals including iron, manganese or phosphorus, or recovery of elevated combined radium/radon products. Radon concentrations in the recovered water did not exceed current regulatory standards but might be a concern if the recovered water were to be utilized to meet ecological demands such as the case of the Everglades ASR program (USACE and SFWMD, 1999). None of these sites reported geochemical issues that hampered site operations. Two of the 15 sites reported major geochemical problems.

The Lytchett Minster ASR pilot project was abandoned due to high concentrations of fluoride in the recovered water (Gaus et al. 2002). The ASR project in Green Bay, Wisconsin was also abandoned due to geochemical constraints related to recovery of heavy metals such as arsenic, nickel, manganese and cobalt (CH₂M Hill, 2003a). The geochemical problems at Green Bay were attributed to oxidation of pyrite minerals located within the aquifer matrix.

A majority of the sites reviewed reported highly turbid or highly colored water upon start of ASR recovery operations. A portion of the turbidity reported is undoubtedly due to build up of TSS in the aquifer or well gravel pack. The remaining cause of the high turbidity or color may be due in part to bacterial growth in the aquifer. Two of the sites reviewed, Seattle and Salt Lake City, reported occasional problems with algae contained within the source water. High algae counts are detrimental to ASR recharge operations since they can result in rapid well clogging.

Besides the hydraulic and water quality considerations, institutional constraints and cost limitations were important factors in several of the ASR sites reviewed. The ASR project in Washoe County, Nevada cited limitations on availability of surface water due to restricted water rights (Bureau of Reclamation, 1996a) as a major project constraint that made the project infeasible. The Salt Lake City project reported that legal and institutional permitting problems resulted in substantial unplanned costs during the ASR test project completed there (Bureau of Reclamation, 1996b).

Geotechnical issues or constraints were investigated at four of the 20 sites reviewed. The Highlands Ranch reported possible hydraulic fracturing of the storage bedrock while subsidence of clay confining units caused radial cracks to form around ASR wells at the

Washoe County project. Both the Seattle and Portland ASR projects investigated operational constraints related to geotechnical engineering issues including subsidence of overlying clay confining units.

Unit cost information was reported or estimated for 13 of the 20 sites.

Development and operation and maintenance costs for the thirteen ASR sites varied widely with a range from \$0.09 to \$2.45 per thousand gallons water recovered, with a geometric mean unit price of \$0.42. Limitations on water availability required the Denver Basin ASR project to purchase water for \$1.65 per thousand gallons (Bureau of Reclamation, 1997).

Brackish water ASR sites

The brackish water ASR sites were located in the United States, India, and Australia. Thirty (30) representative sites were chosen where basic information was available and operational summaries existed or could be easily developed for this report. Generally, the 30 brackish water sites included did not have as much available information as compared to the non-brackish water project sites. In addition, most of the Florida sites were lumped together as they are fairly similar in hydrogeology as well as operational function.

The surveyed ASR projects are located in three countries and four states within the USA. The geology varies across the sites and includes both bedrock (26 sites) as well as unconsolidated sediments (4 sites). Of the bedrock sites, the predominant rock type is sedimentary (all sites) type. All of the sedimentary rock sites are composed of carbonate sequences including limestone, sandy limestone or dolomite. For the case of the four sites situated in unconsolidated geologic environments, sands and gravels dominated the composition of the sediments. Of the 30 sites reviewed, 28 of them had the ASR wells

completed within a confined aquifer while two sites utilized an unconfined aquifer as the ASR storage zone. Of the confined aquifer sites, the storage coefficient was estimated for 21 sites. The storage coefficient for the ASR sites ranged from 2.25×10^{-5} to 4.0×10^{-4} with a geometric mean of 1.32×10^{-4} .

Aquifer transmissivity data were available for 29 out of the 30 sites reviewed. The aquifer transmissivities for the ASR sites ranged from 161 ft²/day to 210,000 ft²/day with a majority (16 of 29 sites where data were available) having values less than or equal to 10,000. The geometric mean transmissivity for the sites was calculated to be approximately 7,698 ft²/day. All of the sites underwent some form of pilot testing during an early stage of the project.

Water quality issues were important at six of the thirty sites reviewed. Disinfection by-products including TTHMs and HAAs were monitored at 15 of the 30 sites reviewed. Of the 15 sites studied, only one site (Boynton Beach) had reported problems with DBP formation in-situ. The remaining sites reported DBP levels declined in ASR storage due to mixing, hydrodynamic dispersion, and biodegradation. Only four sites noted concentrations of DBPs that occasionally exceeded regulatory standards of the United States. This is not unexpected as the reported residual chlorine for four of the sites was relatively low ranging from 0.2 to 4.0 mg/l. A data review of several sites (Charleston and Palm Bay) indicated possible biodegradation of DBPs, although the data are not as definitive as similar sites noted for the non-brackish water projects. Generally for these sites, the level of HAAs was noted as declining more rapidly than associated TTHMs. Tampa and Palm Bay reported that of the TTHM monitored or evaluated, chloroform was the most resistant to biodegradation.

Well clogging issues were monitored at 14 of the 30 sites reviewed. Of these 14 sites, nine reported little to no well clogging issues at all. Three of the 14 sites reported moderate well clogging that required regularly scheduled well back flushing or redevelopment. Of the 14 sites reviewed, only two sites (Lee County and Norfolk) reported a major loss in well recharge or recovery efficiency. The cause of the loss was different for each site. For the Lee County site, high TSS values combined with low aquifer transmissivity ($800 \text{ ft}^2/\text{day}$), led to severe clogging of the carbonate storage zone (Fitzpatrick, 1986). For the Norfolk site, dispersion of illite and montmorillinite clays led to differential clogging of the well gravel pack (Brown and Silvey, 1977). Suspended solids levels were reported at 17 of the 30 sites. Three of the sites reported average TSS concentrations of less than 1.0 mg/l while eight of the sites reported averages of between 1.0 and 5.0 mg/l . Six of the sites reported TSS influent levels greater than 5.0 mg/l including Andrews Farm which experienced TSS levels as high as 170 mg/l for short durations (Barry et al., 2002). Interestingly, all five of the sites that reported moderate to severe well clogging issues, also reported average TSS influent concentrations of greater than 1.0 mg/l . It is important to note that only three of the five sites with moderate to severe well clogging issues had ASR wells completed in carbonate aquifers, while the other two sites had wells screened in unconsolidated sands and gravels. Therefore, of the 12 carbonate sites that were reviewed, only 25% (3 sites) reported clogging difficulties.

The loss of well efficiency due to viscosity differences between the colder recharge water and the warmer ambient groundwater was reported at two of the project sites (Ankeny and Lee County). Five of the 30 sites reviewed reported problems with upconing of more highly brackish water from deeper aquifers. The Cocoa Beach project

site minimized this problem through judicious well back plugging (Pyne, 1995). The Tampa project utilized monitoring and site operations as a way to minimize upconing potential (McNeal, 2004). Two sites reported decreased cycle recovery efficiency due to the existence of conduit flow zones. Well R-2 at the Cocoa Beach ASR project has continually had problems with poor performance due to karst zones or conduits (Pyne, 1995). The same issue was raised at the Willunga ASR project in Australia (Sibenaler et al. 2002).

Geochemical issues were reported on, and evaluated at 16 of the 30 sites reviewed. Of the 16 sites where data were available, eight of the sites reported none or minor geochemical issues. Generally, minor geochemical issues noted included low levels of calcite dissolution, minor enrichment of metals including iron, manganese or phosphorus, or recovery of elevated combined radium/radon products. Radon concentrations in the recovered water did not exceed current regulatory standards but might be a concern if the recovered water were to be utilized to meet ecological demands such as the case of the Everglades ASR program (USACE and SFWMD, 1999). None of these sites reported geochemical issues that hampered site operations. Two of the 16 sites reported moderate geochemical problems related to site operations including influent pH control. The Chesapeake site had moderate problems with excess iron and manganese in the recovered water. The pH of the influent water was slightly acidic and released iron and manganese from the unconsolidated sediments that contained some pyrite. This problem was addressed through influent pH control. By elevating the influent pH to a more highly alkaline range, the iron and manganese stays sequestered in the aquifer. Johnson et al., 1998, discuss some of these same issues in detail. The Hialeah site reported periodic well

clogging by iron hydroxide precipitates due to pH and dissolved oxygen (Reese, 2002). Three of the 15 sites reported major geochemical problems.

The ASR projects at Lee County Olga, Tampa, and Peace River, all have reported significant problems with arsenic contained in the recovered water. The arsenic levels have been as high as 100 ug/l in the recovered water. Luckily, all three sites are able to utilize a combination of water blending and water treatment to reduce the arsenic to acceptable levels for water distribution purposes. Although the arsenic problem seems under some operational control at the three sites, the difficulties have forced the Peace River/Manasota Regional Water Supply Authority to abandon plans for another expansion of ASR at the site (McNeal, 2004).

A majority of the sites reviewed reported highly turbid, highly colored or high-level BOD water upon start of ASR recovery operations. A portion of the turbidity reported is undoubtedly due to build up of TSS in the aquifer or well gravel pack. The remaining cause of the high turbidity or color may be due in part to bacterial growth in the aquifer. Due to these problems, several of the sites instituted an operational change that allows for initial ASR recovered water to be discharged to waste. This minimizes turbidity issues for the recovered water. The Lee County site reported occasional problems with algae contained within the source water. High algae counts are detrimental to ASR recharge operations since they can result in rapid well clogging.

Besides the hydraulic and water quality considerations, cost limitations were important factors in several of the ASR sites reviewed. Based upon the review of documents available to the author, none of the 30 sites reviewed addressed potential geotechnical issues such as hydraulic fracturing or subsidence of clay confining units.

Unit cost information was reported or estimated for five of the 30 sites.

Development and operation and maintenance costs for the five ASR sites varied with a range from \$0.41 to \$2.35 per thousand gallons water recovered, with a geometric mean unit price of \$0.94.

Development of ASR simulator models

A number of investigators have undertaken modeling of ASR systems in order to explore assorted controls on system performance. Khanal (1980), Merritt (1986), Huntley and Bottcher (1997), Merritt (1997), Wright and Barker (2001), Missimer et al. (2002), and Pavelic et al (2002), have all developed models of brackish-water ASR projects. Khanal and Pavelic et al. utilized conventional two-dimensional models to explore the expected performance. Merritt and Missimer et al. utilized fully-density dependent three-dimensional models to evaluate realistic operational scenarios and the effects of various independent variables (porosity, vertical hydraulic conductivity, density of ambient groundwater, etc.) upon system performance. Pavelic elected not to utilize a fully-density dependent model to develop similar performance estimates. Streetly (1998) and Anderson & Lowry (2004) have investigated ASR performance in near-potable aquifers using numerical models. Sedighi (2003) developed a semi-analytical approach for estimating ASR cycle testing performance using stochastic variables.

As part of this research effort, a family of ASR simulator models was developed to confirm and evaluate ASR performance variables. The models were all originally constructed utilizing the United States Geological Survey (USGS) groundwater flow model code MODFLOW (McDonald and Harbaugh, 1988) and the contaminant transport code MT3DMS (Zheng and Wang, 1999). One of the models was also tested using the density-dependent code SEAWAT (Guo and Langevin, 2002). SEAWAT utilizes native

MODFLOW and MT3DMS files to calculate the groundwater flow field in cases where density of the groundwater is not constant. Due to long model run times, SEAWAT was only utilized for a limited number of simulation cases. The SEAWAT simulations were part of an ongoing cooperative research effort shared between the U.S. Army Corps of Engineers and the USGS.

Four separate models (A, B, C, and D) were constructed as part of this research effort. First, a fully three-dimensional model was constructed to simulate ASR performance in a confined aquifer (Simulator A). The initial three-dimensional model was based upon the hydrogeology of Palm Beach County, Florida. The U.S. Army Corps of Engineers has developed a series of ASR test projects throughout south Florida (USACE and SFWMD, 2004). The Hillsboro ASR Pilot Project is one of these sites. It is located in south-central Palm Beach County. The project site has been fully investigated through the installation of test wells, aquifer testing, and groundwater sampling. Therefore, the hydrogeologic parameters of this area served as the basis for the base ASR simulator model. The three-dimensional model contains 16 layers representing the Surficial Aquifer System, the Hawthorn Group confining unit, and the Floridan Aquifer System (FAS). The proposed ASR storage zone of the Hillsboro site is within the upper portions of the FAS. The FAS zone includes four equal model layers for Simulator A.

Due to the large recharge/recovery rate anticipated for the Hillsboro pilot test, the model boundary included a large part of Palm Beach County, Florida USA. The model grid is 40 miles X 40 miles and consists of an irregularly spaced grid with 10 feet grid resolution at the ASR well location and 3,000 feet resolution at the model boundary. The

model has a total of approximately 185,000 grid cells. Initially, smaller domains were tested to determine the artificial influences induced by the boundary conditions; however, all smaller domains tested introduced significant induced error. The three-dimensional grid allowed for exploration of many ASR performance issues including the effects of vertical leakage between aquifers. Although the Hillsboro hydrogeology was replicated for initial model simulations, a matrix of hydrogeologic properties was investigated using the model as a “prototype-testing model”. Using the model as a prototype, various ASR performance parameters were investigated. This approach is consistent with work completed by Merritt (1986) and Streetly (1998). The three-dimensional grid was also imported into SEAWAT to evaluate density effects on ASR performance (Simulator B).⁶ Unfortunately, due to the model complexity, the run times for the various ASR simulations were long, ranging from several hours to 14 hours. Figure 2-24 displays the model grid for ASR Model Simulator A. In addition to the base three-dimensional model and the SEAWAT version of the model, two additional confined aquifer models were constructed.

The two additional models included more horizontal resolution but less vertical resolution. ASR Simulator model C contained one vertical layer with a horizontal resolution of five feet at the ASR well. The model domain was the same as ASR Simulators A and B but had 167 rows and 167 columns for approximately 28,000 grid cells. ASR Simulator model D also contained one vertical layer but had even more horizontal resolution. Simulator model D had the same domain but utilized a two feet resolution at the ASR well. Model D contained 250 rows and 250 columns for

⁶ The original SEAWAT model simulator B was developed by the United States Geological Survey Miami office

approximately 62,500 grid cells. Both of these models were extremely accurate when compared to existing analytical models, however, ASR Simulator C was utilized more often due to shorter run times.

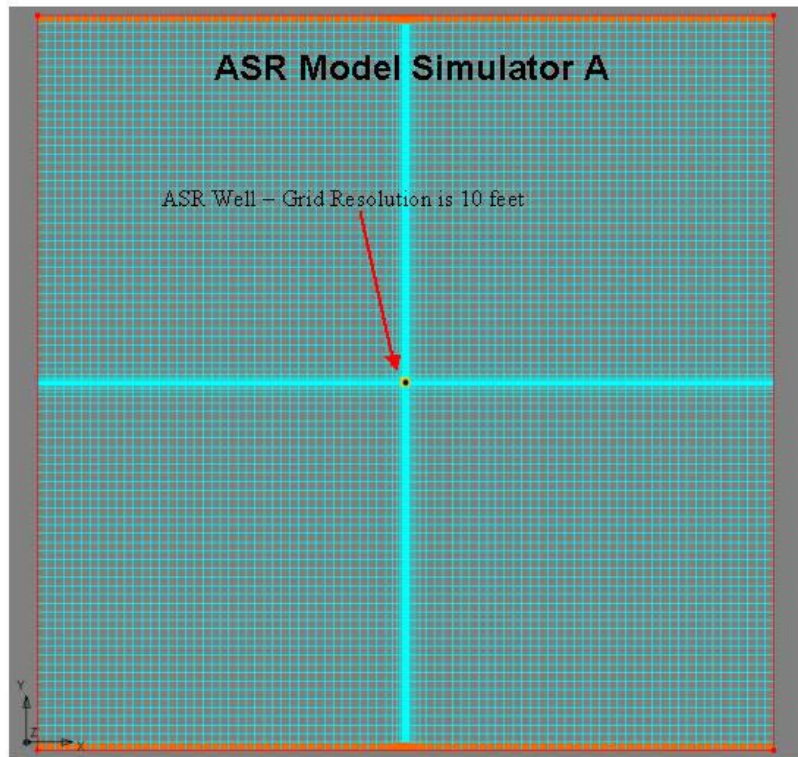


Figure 2-24. Model domain for ASR model simulator A, the domain is 40 miles

Figure 2-25 depicts a comparison of numerical model steady-state simulation results and analytical simulation results. The simulation compares drawup (aquifer mounding as denoted by “s”) caused by a well injecting into a confined aquifer. Both the numerical model (Simulator C) and the analytical solution predict similar mounding effects, especially close to the ASR well itself.

The whole family of simulator models was utilized as part of this research effort, with simulators A and C being utilized for the largest number of model runs. Over 800 separate simulations were completed for this study. In addition to comparisons of

numerical results to analytical solutions, intra-model comparisons were completed using different contaminant transport solution methods.

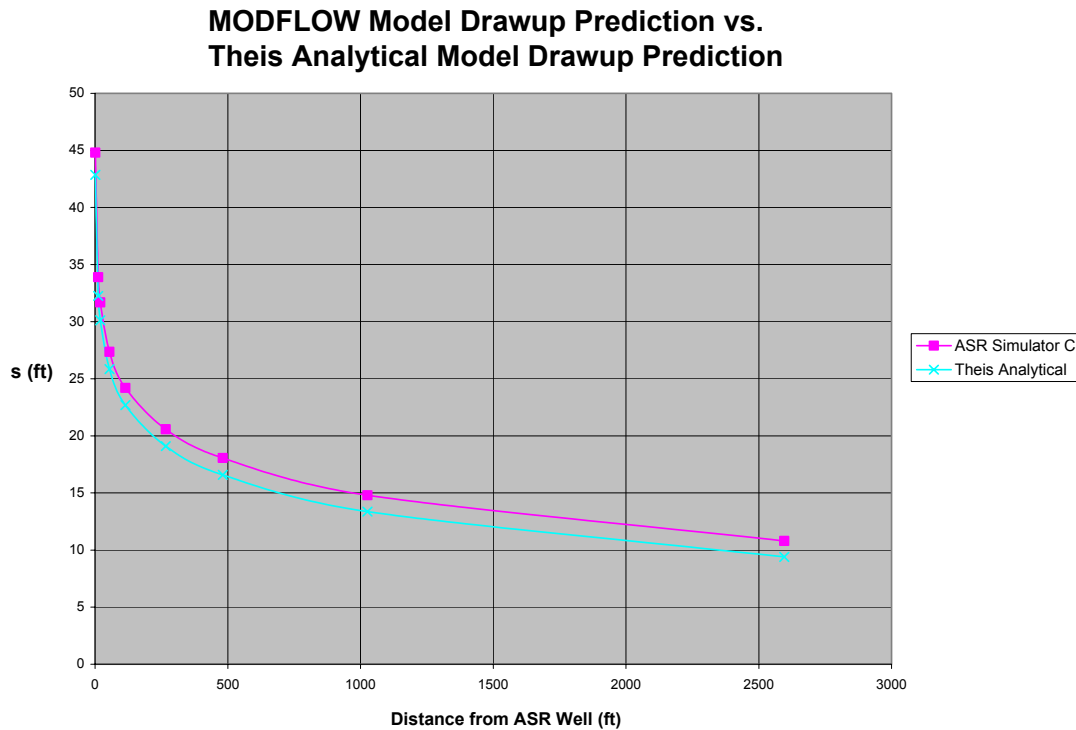


Figure 2-25. Comparison of model predictions with numerical and analytical models

Care must be taken during the selection of the horizontal grid resolution to ensure model stability and accuracy. In addition, the various contaminant transport solution algorithms may be subject to errors.

Merritt (1985; 1986) discusses numerical dispersion problems with backward difference techniques like “upwinding”. Merritt notes that the central difference technique greatly reduces numerical dispersion although it can still be significant. Apparent degree of dispersion in the model computations is primarily dependent upon local flow velocities. Central difference techniques may also be subject to oscillatory behavior including undershoot and overshoot. The criteria to control this phenomenon are more restrictive as dispersivity values approach 0.

Merritt (1993) modeled a thin brackish aquifer overlain and underlain by confining layers containing higher salinity water. He found that the estimate of recovery efficiency was strongly influenced by the finite-difference approximation method, the algorithm for representing vertical advective and dispersive fluxes, and the values assigned to parametric coefficients that specify the degree of vertical dispersion and molecular diffusion. Voss and Souza (1987) discuss measures to assess the degree of numerical dispersion and velocity errors in numerical models. Voss and Souza (1987) discuss the error that can be introduced in density-dependent models related to the velocity distribution in both the horizontal and vertical direction. Model resolution is thought to be the most important consideration to minimize potential errors or oscillations in the velocity distribution. Unfortunately, for ASR projects, high velocities result to to well recharge actions. Therefore, fine horizontal and vertical resolution may be required to accurately simulate ASR performance.

Haitjema et al. (2001), discuss numerical inaccuracies in contaminant transport models due to MODFLOW derived velocity field. The accuracy of MODFLOW in representing a ground water flow field determines in part the accuracy of the transport predictions. Woods (2004) discovered during her research efforts that the SUTRA model (Voss and Provost, 2002), a commonly used finite-element model, is numerically dispersive. Wood confirmed this conclusion by using another benchmark, the “Gauss pulse test” (that has an analytical solution), to test SUTRA against.

Figure 2-26 displays the results of various ASR Simulator models along with

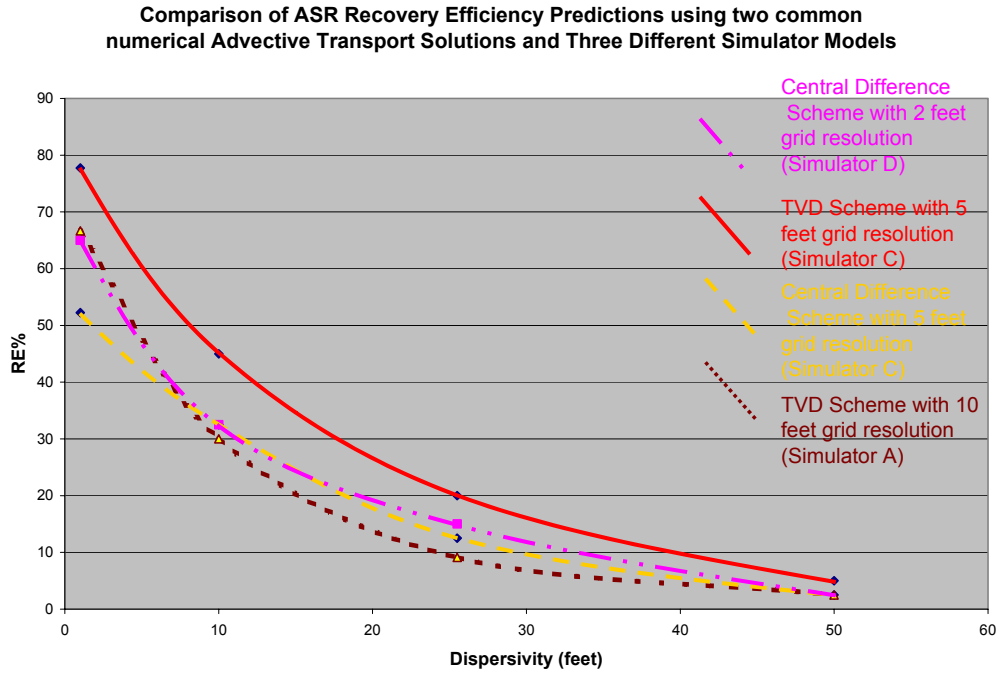


Figure 2-26. Comparison of ASR simulator models and alternate transport solutions different alternate advective transport solutions. As part of this research effort, intra-model comparisons were completed to determine accuracy, model stability, and model run times. The figure plots the predicted RE percentage versus model assigned dispersivity. Note that at low dispersivities, where advection is the dominant form of transport and mixing, solutions subject to numerical dispersion (Central Difference Solution) predict RE estimates lower than codes that have none (TVD Solution). At relatively high values of dispersivity (e.g., 50 feet), the model predictions nearly converge. Similar experiments were completed using ASR Simulator B using the SEAWAT model code. Figure 2-27 displays a comparison simulation with 30 days recharge (@ 5MGD), 305 days of storage, and 30 days of recovery for an ASR storage zone containing 4,000 mg/l TDS and the recharge water containing 150 mg/l TDS.

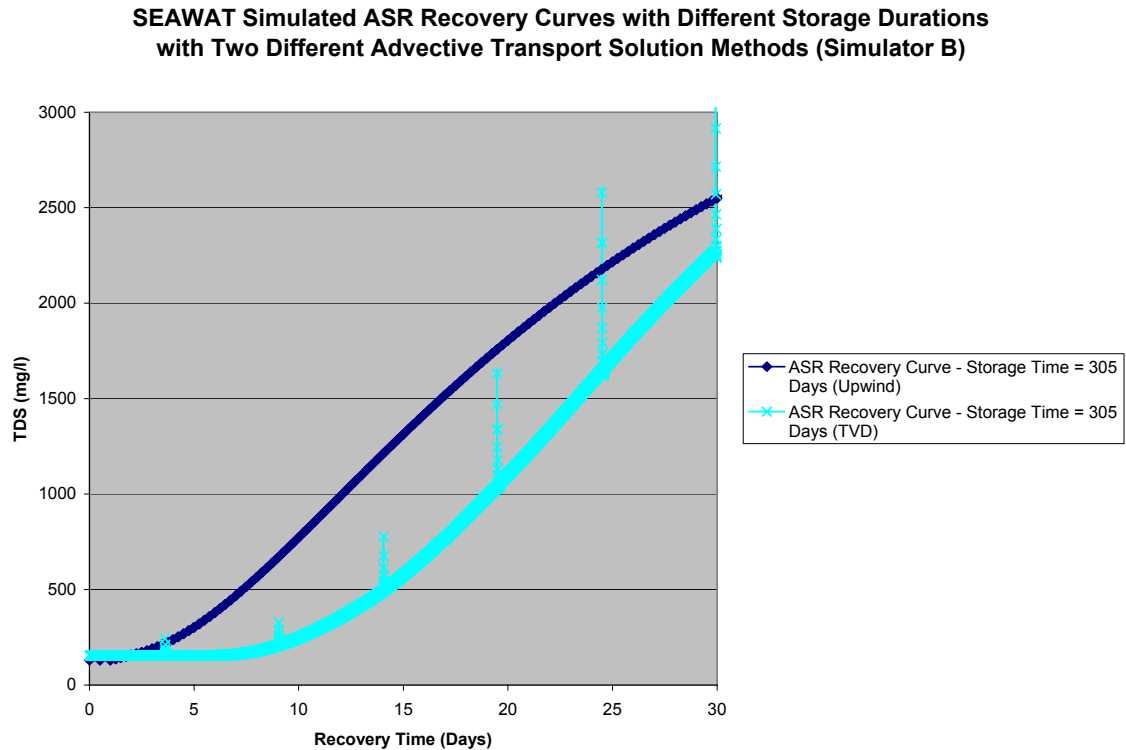


Figure 2-27. Comparison of ASR Simulator B results using two alternate transport solutions

The resulting time versus concentrations graphs are quite different for the upwind solution versus the TVD solution. Minor oscillations observed in the TVD solution did not inhibit the interpretation of model simulation results. Besides model comparisons and comparisons of solutions, comparison of MT3DMS and SEAWAT results was performed to determine the degree of error introduced into RE estimates if density is assumed to be constant. These simulations were important to show that at ambient groundwater TDS values of less than 4,000 mg/l and at zero to minimal storage durations, non-density predictions are practically the same as those including density. As ASR storage durations approach 305 days however, this assumption is no longer valid and a density-dependent model must be utilized to provide RE estimates. Figure 2-28 displays a comparison of MODFLOW/MT3DMS simulation results versus SEAWAT results for

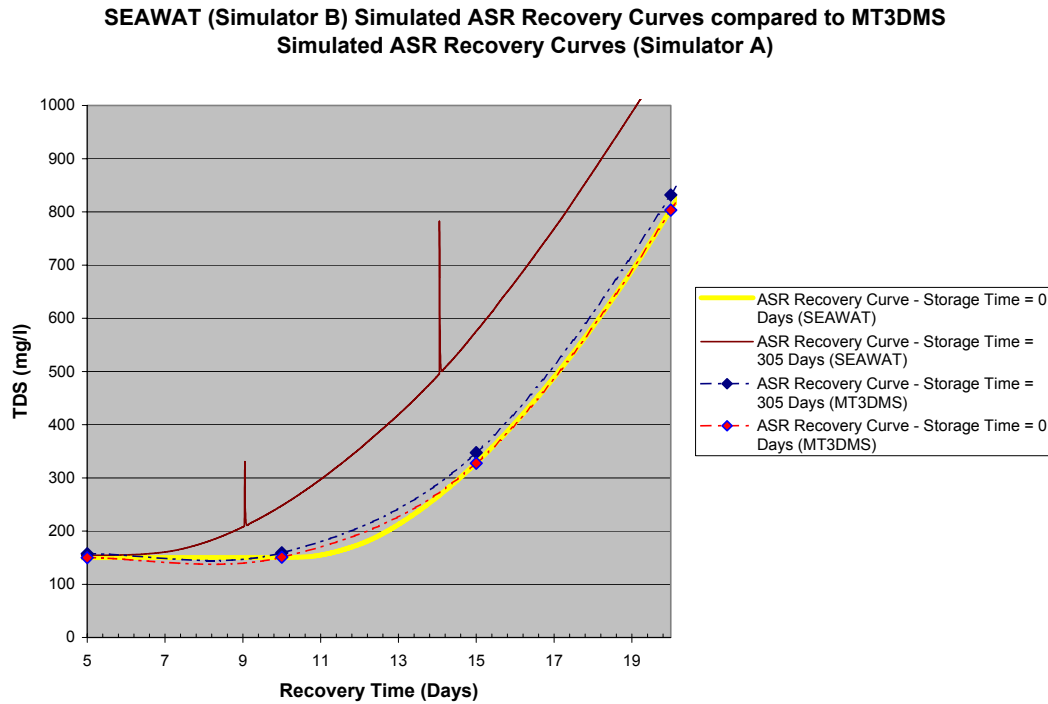


Figure 2-28. Comparison of ASR Simulator A and B results using same model run conditions with ambient groundwater TDS concentration equal to 4,000 mg/l

the same model run conditions. Based upon the intra-model comparison and a comparison of available solution techniques, ASR Simulator models A and C were selected for use to evaluate various ASR performance factors. Simulator C was utilized to compare the effect of different system variables on RE performance, while Simulator A was utilized for development of empirical performance envelopes discussed in Chapter 3 of this report. Simulators A and C were utilized instead of Simulator D due to faster model run times and comparable accuracy. Simulator B (SEAWAT density-dependent model) was utilized in evaluating the effect of storage duration on system performance. For all simulators, the TVD transport solution was selected to minimize numerical dispersion.

Confirmation of ASR performance factors

As described previously in this report, many variables can affect ASR performance. Previous authors including Khanal (1980), Merritt (1986), Yobbi (1996), Streetly (1998), Missimer et al. (2002), Pavelic et al. (2002), and Anderson and Lowry (2004) have studied the effect of important variables on ASR RE percentage. Generally, each author has evaluated ASR performance based upon the initial cycle RE percentage. For this report, both the initial RE percentage as well as the long-term RE was evaluated. For this research effort, the following variables were assessed:

- Thickness of storage zone
- Effect of aquifer regional pre-existing gradient
- Recharge volume
- Dispersivity
- ASR storage duration
- Density of ambient groundwater in storage zone
- Ambient groundwater quality
- Effect of multiple consecutive cycles
- Aquifer homogeneity
- Transmissivity
- Unequal recharge and recovery rates
- Degree of anisotropy
- Porosity

Of the variables investigated, the last four listed indicated little effect upon ASR performance as measured by RE percentage. Aquifer transmissivity, degree of anisotropy, and unequal recharge and recovery rates did not affect the model predicted

cycle one RE percentage. Porosity was generally insensitive in a reasonable range of assigned values from 25 to 50%. At extremely low porosity values (e.g. 5%), RE percentage declined by 10 to 12%.

Using Simulator C, the effect of storage zone thickness on RE percentage was investigated. Using storage zone thickness of 50 feet, 100 feet, and 200 feet, multiple simulations were run to evaluate performance. For all model runs, the ambient groundwater chloride concentration (AWQ) was assigned to be 1,000 mg/l and the recharge water chloride concentration (RWQ) was assigned to be 5 mg/l. Also, the recharge rate was assigned to be 0.5 MGD for 30 days. Recovery was commenced immediately after recharge was completed and was 0.5 MGD for a total duration of 30 days. The recovery ended once the concentration of chloride exceeded 250 mg/l. The RE was defined as this recovered volume divided by the volume recharged as a percentage. In addition, these simulations were run using different values of dispersivity. Figure 2-29 depicts the simulation results. It is evident from a review of the results that thinner storage zones provide better performance, however, the difference is more evident as the dispersivity increases.

Interestingly, at very high dispersivity values, the performance curves appear to approach each other again. This may suggest some optimum combination of aquifer thickness and dispersivity could be determined at a prospective project site. This finding is consistent with previous researchers including Merritt (1986) and Kimbler et al. (1975).

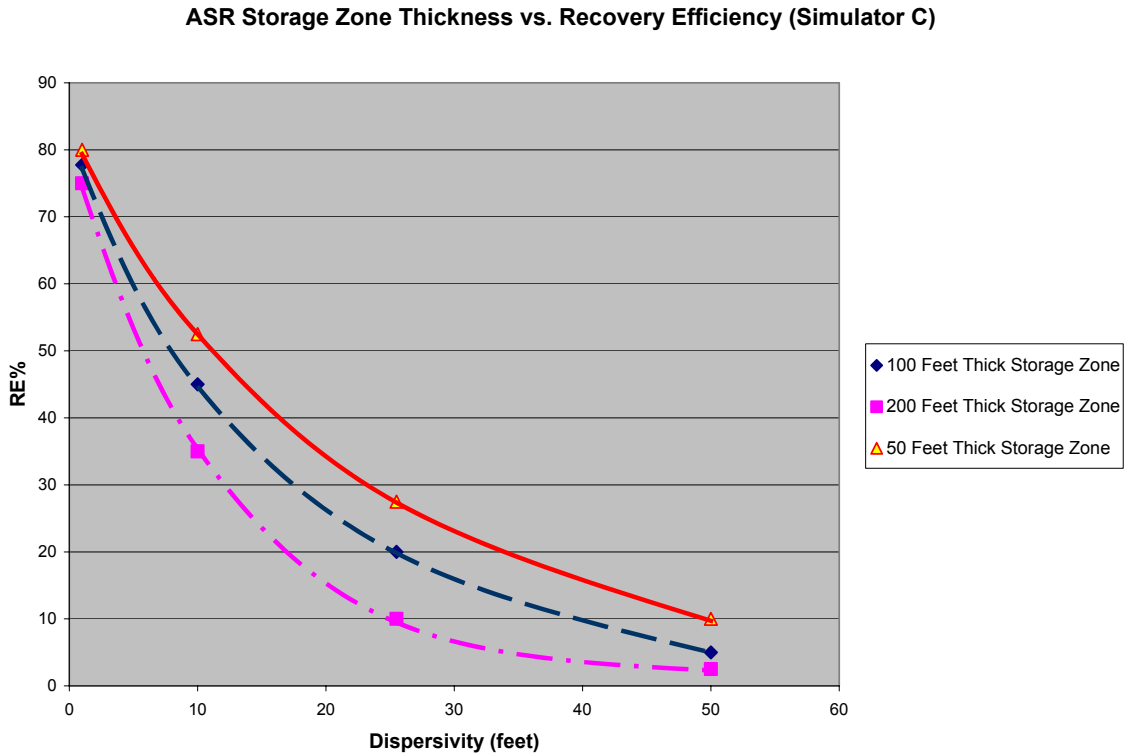


Figure 2-29. Comparison of ASR simulator C predicted RE% results using same model run conditions with AWQ equal to 1,000 mg/l and RWQ equal to 5 mg/l, but changing the storage zone thickness

Using Simulator C, the effect of a pre-existing groundwater gradient on RE percentage was investigated. Initially, it was assumed that a pre-existing gradient would translate the recharged bubble of freshwater down gradient resulting in a small amount of unrecoverable water. Simulation conditions were the same as the previous case discussed for Figure 2-29, namely the AWQ was equal to 1,000 mg/l chloride and the RWQ was equal to 5 mg/l chloride. Also the recharge rate and duration were the same as previous simulations. The simulation set was also run using two different dispersivities including one foot and ten feet. Figure 2-30 depicts the results of simulations evaluating this possibility.

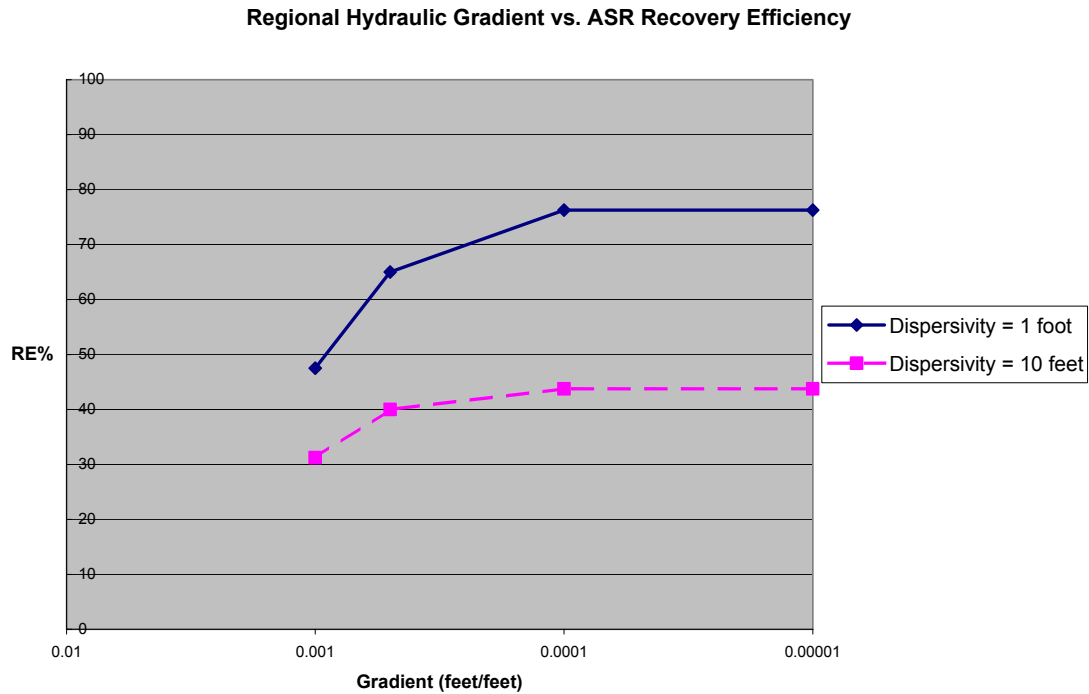


Figure 2-30. Comparison of ASR simulator C predicted RE% results using same model run conditions with AWQ equal to 1,000 mg/l and RWQ equal to 5 mg/l, but changing the regional pre-existing groundwater gradient

The figure clearly shows the effect of a regional pre-existing gradient on RE%. For cases of low dispersion in the aquifer (e.g., dispersivity equal to 1 foot), the performance is quite sensitive to the gradient once it is greater than 0.0001. At a dispersivity of 10 feet, the effect of regional gradient is less pronounced. This outcome is consistent since at low values of dispersivity, advective transport is dominant so that the gradient is much more important than at comparable cases where the dispersivity is higher. In cases of large dispersion, the contaminant transport algorithms include both advective and dispersive components.

Using Simulator C, the effect of recharge volume on the initial RE percentage was investigated. The model conditions were identical to the previous two sensitivity evaluations except that the recharge volume was continually increased to approximately

40 million cubic feet (300 million gallons recharged). Also, similar to previous comparisons, different dispersivities were also evaluated along with recharge volume. Three different dispersivities (alpha) including one foot, ten feet, and 25.5 feet were investigated. Figure 2-31 depicts the results of the model simulations.

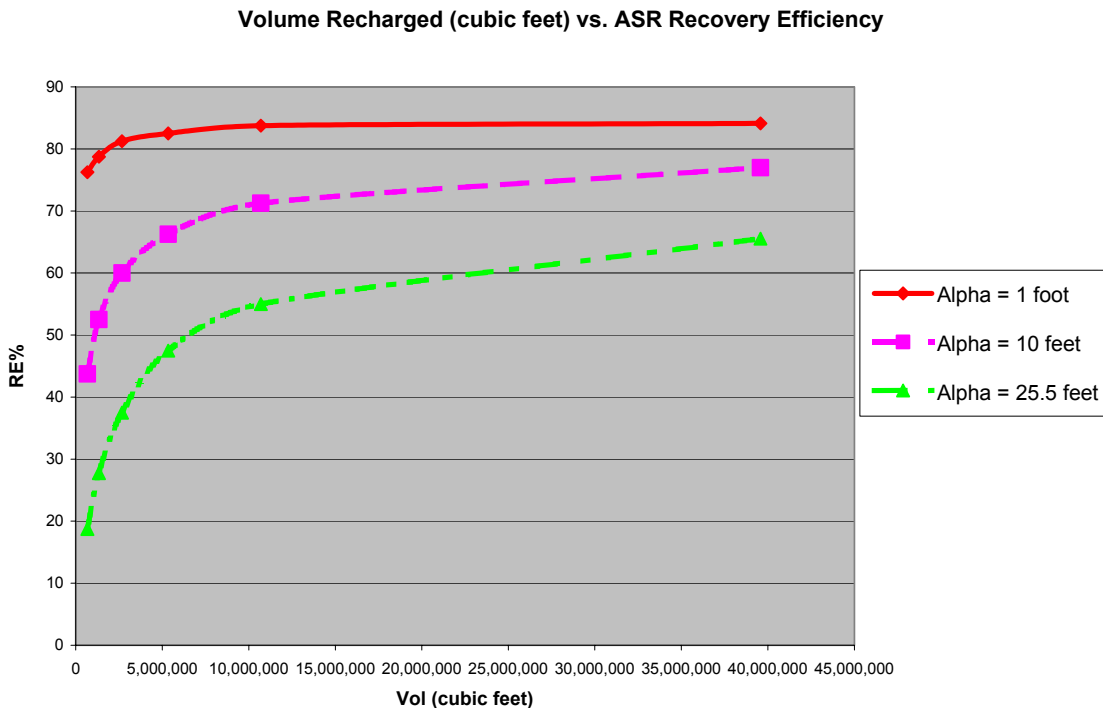


Figure 2-31. Comparison of ASR simulator C predicted RE% results using same model run conditions with AWQ equal to 1,000 mg/l and RWQ equal to 5 mg/l, but changing the recharge volume

From a review of the figure, it is evident that increasing recharge volume does improve ASR performance initially but eventually reaches a point of diminishing returns. For all three dispersivity values, the improvement of RE percentage beyond 10 million cubic feet (75 million gallons) of recharge is minimal. Beyond this value, the improvements are slight and approach an asymptote. Interestingly, at low dispersivity values, the asymptote is reached quickly while at higher values of dispersivity the asymptote is not completely reached at 40 million cubic feet of recharge.

Using Simulator C, the effect of dispersivity on the initial RE percentage was investigated. The model conditions were identical to the previous evaluations except that four separate assigned values of dispersivity were evaluated including one foot, ten feet, 25.5 feet, and 50 feet. Figure 2-32 depicts the results of the model simulations.

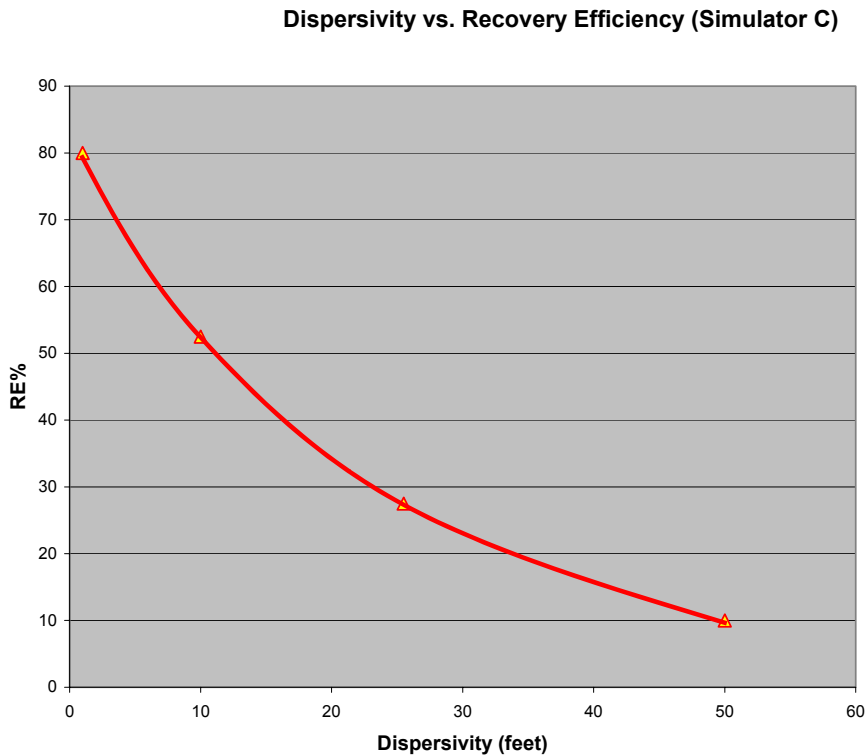


Figure 2-32. Comparison of ASR simulator C predicted RE% results using same model run conditions with AWQ equal to 1,000 mg/l and RWQ equal to 5 mg/l, but changing the dispersivity

The model simulations reveal that the amount of dispersion that occurs within the ASR storage zone is critical. The RE percentage at a dispersivity of one foot is 78% compared to a RE percentage of 10% at a dispersivity of 50 feet. It is also important to note that the predicted RE percentage at a dispersivity of 25.5 feet is 27.5%, as compared to a value of 44% if the effect of dispersivity on RE percentage were a linear relationship. In actuality, curve fitting using least-squares methodology showed that the relationship is logarithmic in nature. A best-fit logarithmic curve is shown on Figure 2-33 along with

the correlation coefficient and curve equation. Although the fit is not perfect, the r^2 coefficient is greater than 95% suggesting a significant relationship.

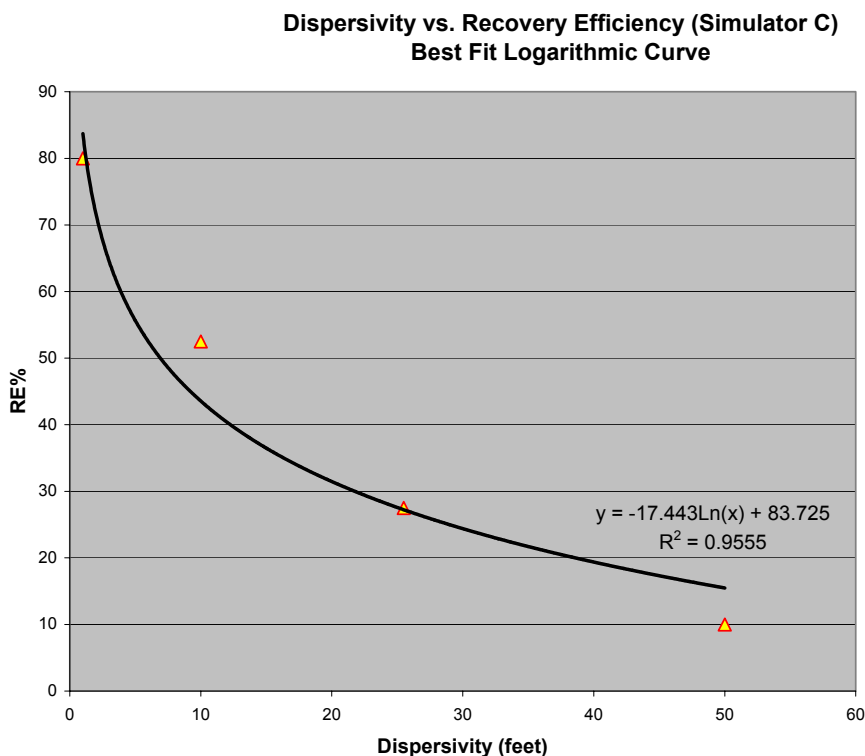


Figure 2-33. Comparison of ASR simulator C predicted RE% results using same model run conditions with AWQ equal to 1,000 mg/l and RWQ equal to 5 mg/l, showing best-fit logarithmic curve

With help from the USGS Miami office, Simulator B was utilized to evaluate the effect of storage time on the initial RE percentage.⁷ The model conditions for these simulations were different than previous cases. For this comparison, SEAWAT was utilized since density effects were found to be important for longer storage durations. For the comparisons, the ASR storage zone was assigned an initial concentration of 4,000 mg/l TDS, the recharge water had an assigned TDS value of 150 mg/l, the recharge and recovery rate was five MGD, and the dispersivity was assigned a value of zero. For each

⁷ Email of model results and output files from Christian Langevin and Mike Zygnerski from USGS Miami

simulation, 30 days of recharge was followed by 30 days of recovery. For the first simulation, recovery commenced immediately following recharge. For the second simulation, recovery commenced following 30 days of storage. For the last simulation, recovery commenced following 305 days of storage. Figure 2-34 shows the ASR recovery curve for each simulation.

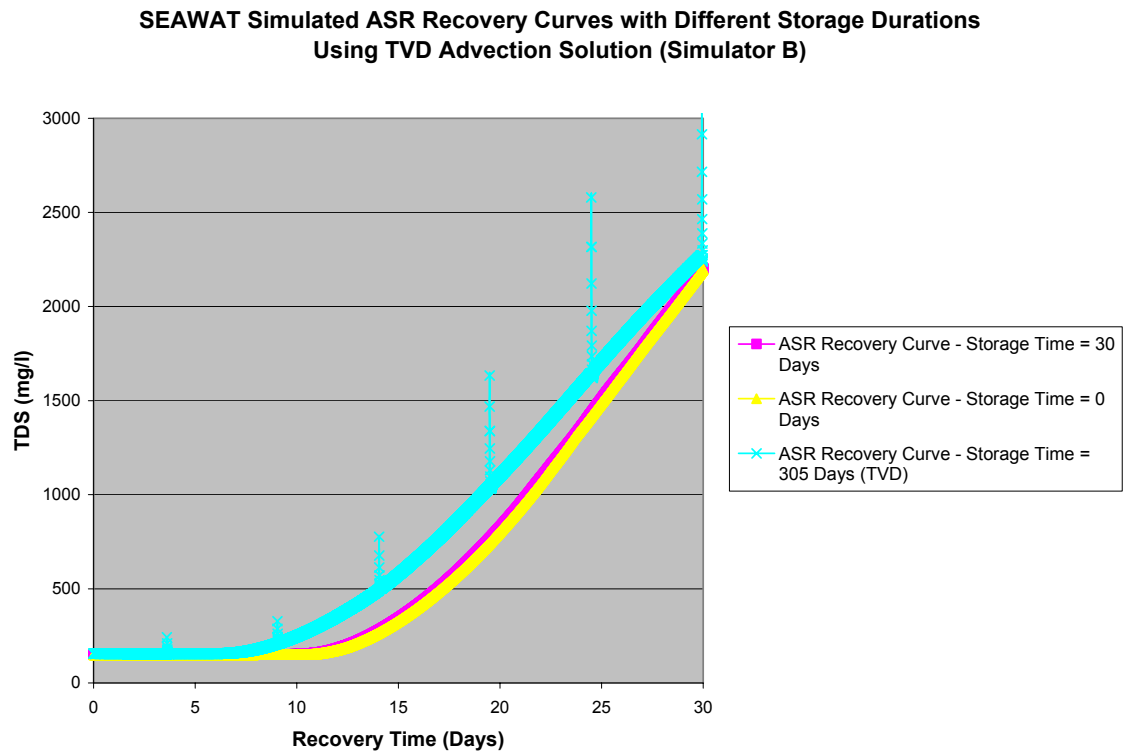


Figure 2-34. Comparison of ASR simulator B predicted RE% results using same model run conditions with ambient groundwater TDS equal to 4,000 mg/l and recharge TDS equal to 150 mg/l, dispersivity is assigned zero feet

The figure clearly demonstrates that long storage times do reduce ASR RE percentage. For no storage duration the RE percentage for this case was 57.21%. With 30 days of storage the predicted RE percentage was 56.51%, only a slight reduction in system performance. At a storage duration of 305 days the predicted RE percentage was 47.17%, a performance reduction of approximately 18% as compared to the base case with zero days of storage. Also, using both Simulator A and B for these simulations has

shown that a majority of the performance reduction is attributable to buoyancy stratification from the density differential between the ambient groundwater and the recharge water.

With assistance from the USGS Miami office, Simulator B was utilized to evaluate the effect of density differential on RE percentage. Initially, the density of the ASR storage zone was based upon a TDS value of 4,000 mg/l. As explained in the previous section, the effect of density differentials of similar magnitude is typically negligible, however, as the ambient groundwater TDS value approaches that of seawater (e.g., 35,000 mg/l), the effect of density becomes quite pronounced. Using a seawater storage zone and storage durations of zero days, thirty days and 305 days as the previous evaluation, and assigning a dispersivity of zero, the predicted RE percentage for all simulations was zero to five percent. Obviously in a real aquifer, performance might be even worse due to dispersion effects. Figure 2-35 depicts the simulated ASR recovery curve for the case of 305 days of storage. The figure includes results using both the TVD and upwinded advection solutions along with results from WASH123, a density-dependent finite-element code where the same simulation was performed for quality control and comparison purposes. The results of the modeling in these instances are very sensitive to both the horizontal and vertical resolution of model geometry. In cases where groundwater quality trends from freshwater to saltwater (e.g., a vertical concentration sharp front gradient) in the vertical direction, Voss and Souza (1987) note that finer resolution is required in the vertical direction than in the horizontal direction. Voss and Souza (1987) also recommend limiting the mesh or grid horizontal resolution to less than four times the longitudinal dispersivity.

**Simulator B - Storage of freshwater in Seawater Aquifer Model Comparison
Concentration Vs. Time Curve from Simulations with Dispersivity = 0
Average Concentrations from all Layers**

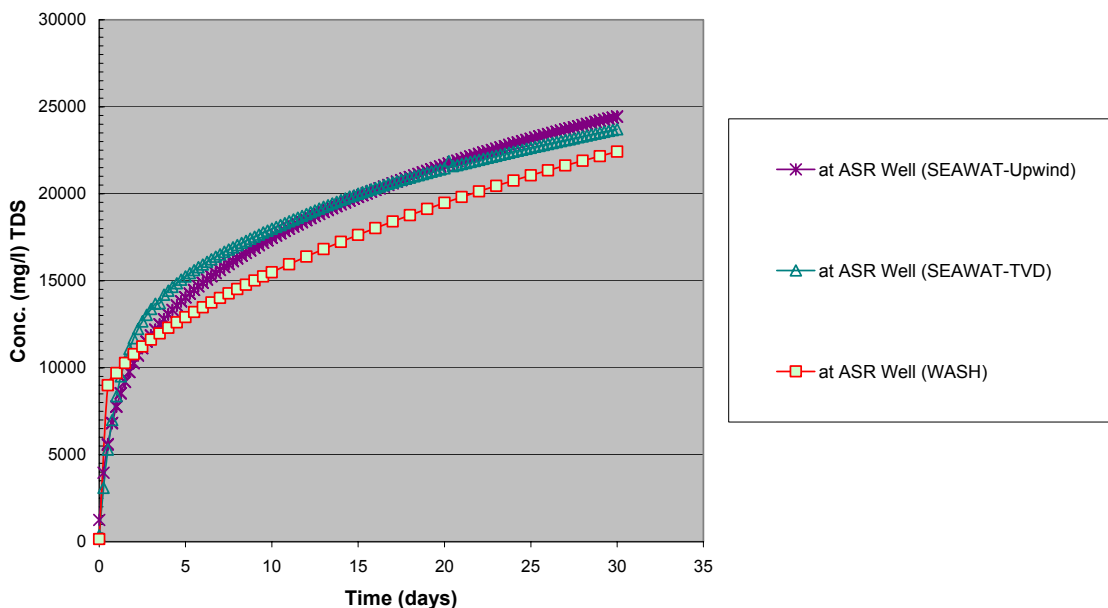


Figure 2-35. Comparison of ASR simulator B predicted RE% results using ambient groundwater TDS equal to 35,000 mg/l and recharge TDS equal to 150 mg/l, dispersivity is assigned zero feet

Using Simulator C, the effect of ambient groundwater quality on RE percentage was investigated. Initially, the simulations utilized an AWQ of 1,000 mg/l chloride and a RWQ of 5 mg/l. Then simulations were completed increasing the AWQ to 2,000 mg/l and then to 3,000 mg/l. All simulations began recovery of freshwater immediately after the completion of recharge operations. All simulations utilized a 5 day period of recharge with a recharge and recovery rate of four MGD. Dispersivities (α) were varied for different simulations in order to develop a family of curves. For each simulation, the recovery period ended once the chloride concentration exceeded the regulatory limit of 250 mg/l. In order to maximize efficiency, these simulations were continued for six separate recharge and recovery cycles in order to also assess the extent

of RE improvement over time. Figure 2-36 depicts the first set of simulations using an AWQ of 1,000 mg/l of chloride.

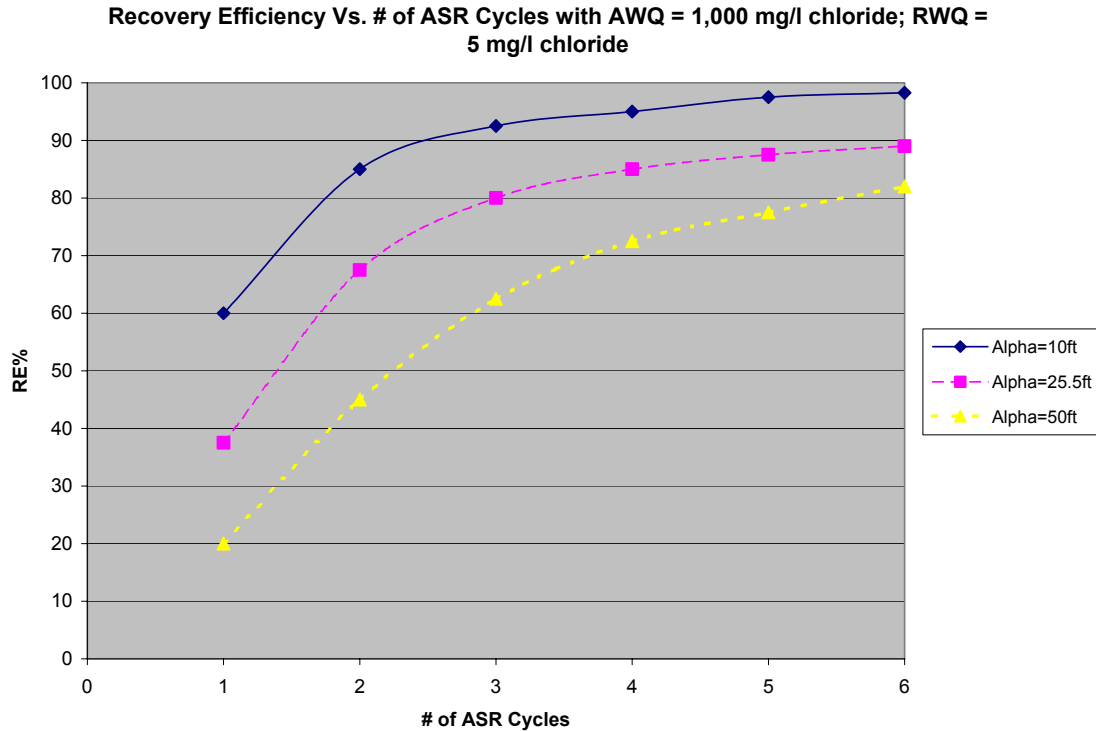


Figure 2-36. Comparison of ASR simulator C predicted RE% results using ambient groundwater chloride equal to 1,000 mg/l and recharge chloride equal to 5 mg/l, dispersivity is assigned as shown

Figure 2-37 depicts the second set of simulations using an AWQ of 2,000 mg/l of chloride. Figure 2-38 depicts the third set of simulations using an AWQ of 3,000 mg/l of chloride. The figures clearly show a number of interesting results. First, as the ambient groundwater chloride increases, the RE percentage obtained for each cycle decreases. As an example, for the case of dispersivity of 10 feet, the RE percentage at the end of cycle 6 declines from 97.5% to 83% as the AWQ rises from 1,000 to 3,000 mg/l. Second, the RE percentage increases as the number of recharge and recovery cycles increases up to an asymptotic value that depends upon dispersivity and AWQ. This phenomenon is due to residual freshwater left in the aquifer after each recovery cycle as long as the recovery

cycle is halted once the regulatory value of 250 mg/l chloride is exceeded. Third, as the ambient groundwater chloride increases, the number of cycles required to reach the maximum RE percentage asymptote increases. As an example, with a dispersivity of 10 feet and an AWQ of 1,000 mg/l chloride, the maximum RE asymptote is reached by cycle 6, whereas, with a dispersivity of 10 feet and an AWQ of 3,000 mg/l chloride, the maximum RE asymptote is not reached by cycle 6. A similar pattern is shown when cumulative recovery efficiency (CRE) percentage is plotted versus the number of cycles. CRE percentage is useful in evaluating long-term ASR performance as well as system cost effectiveness. Figure 2-39 depicts the same information as Figure 2-37 except that CRE is plotted instead of RE. The curves in Figure 2-39 are typically smoother than

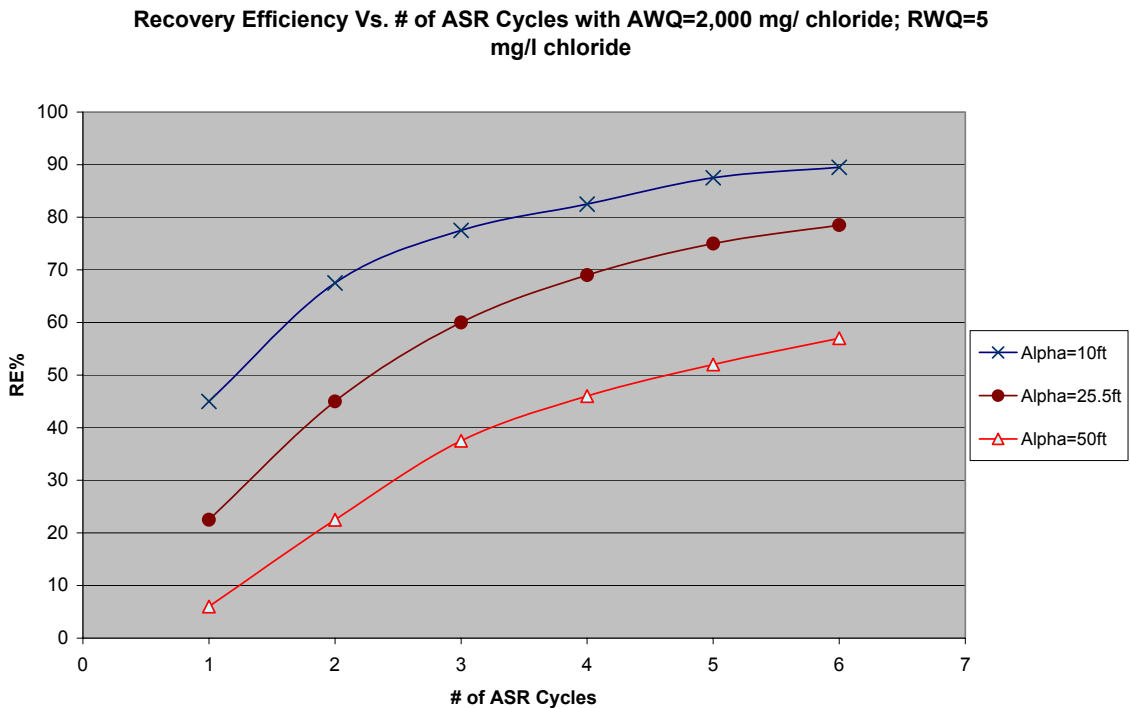


Figure 2-37. Comparison of ASR simulator C predicted RE% results using ambient groundwater chloride equal to 2,000 mg/l and recharge chloride equal to 5 mg/l, dispersivity is assigned as shown

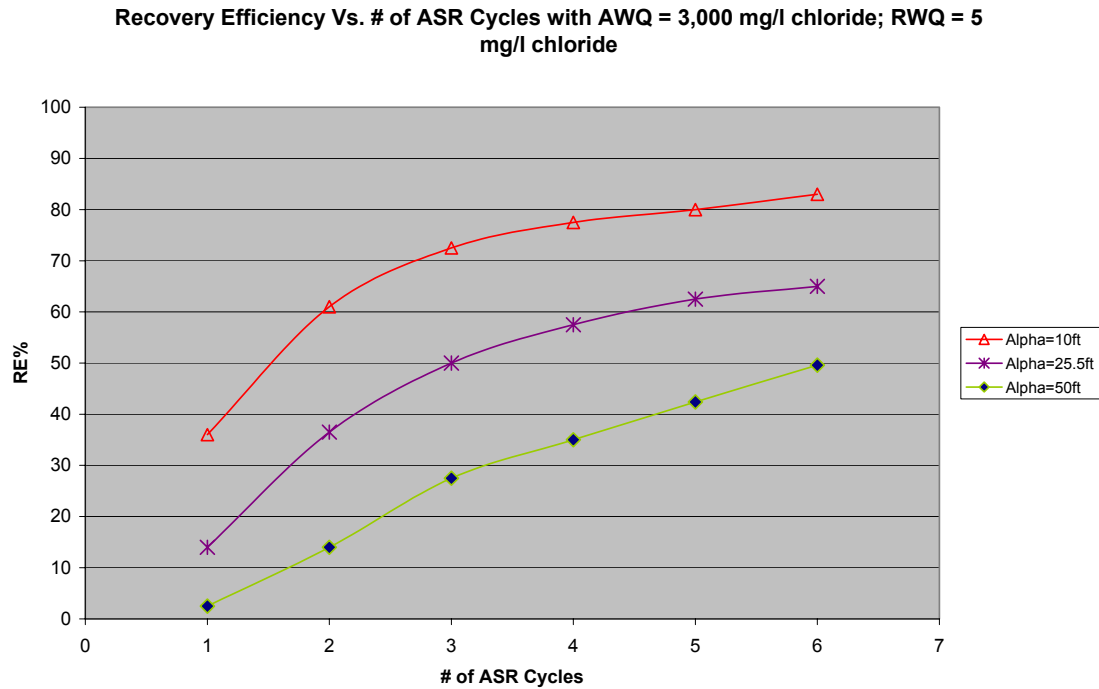


Figure 2-38. Comparison of ASR simulator C predicted RE% results using ambient groundwater chloride equal to 3,000 mg/l and recharge chloride equal to 5 mg/l, dispersivity is assigned as shown

those shown in Figure 2-37 and when curve fitting techniques were utilized, the curves depicted on Figure 2-39 fit a logarithmic model very well. Figure 2-40 depicts the same information on Figure 2-39 with best-fit logarithmic curves matched to the model results. The r^2 coefficient for each curve is greater than 0.95 revealing a highly significant relationship between CRE percentage and # of ASR cycles. This correlation may provide an effective means of predicting future ASR performance after a few initial field cycles are completed at a particular project site. This relationship could be utilized by the ASR community of practice in extrapolating the future performance of operating projects. Also, using the CRE data to estimate future performance may empower ASR users in developing refined operating plans at existing projects. In addition, new performance metrics utilizing CRE percentage could be useful in evaluating ASR project economics.

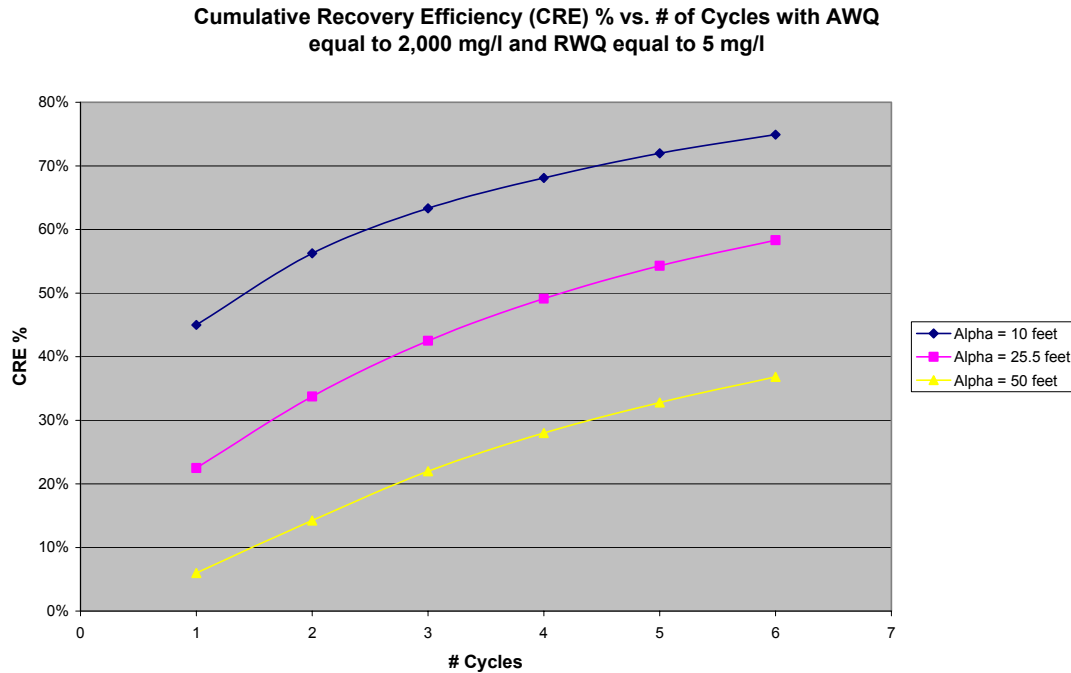


Figure 2-39. Comparison of ASR simulator C predicted CRE% results using ambient groundwater chloride equal to 2,000 mg/l and recharge chloride equal to 5 mg/l, dispersivity is assigned as shown

The final ASR performance factor that was evaluated using the ASR simulator models was a comparison between a perfect homogeneous model and a heterogeneous model. For this comparison, Simulator C was utilized using an AWQ of 1,000 mg/l chloride and a RWQ of 5 mg/l and a dispersivity of 10 feet. First, the homogeneous simulations were completed for a series of six recharge and recovery cycles. Then, random heterogeneity was added to the model transmissivity distribution. Both lower and higher transmissivities were added randomly around the ASR well location. This type of simulation evaluation is useful since real aquifers are more likely to exhibit heterogeneous hydraulic parameter distributions. Also, the simulation reveals the overall effect of heterogeneity in ASR performance.

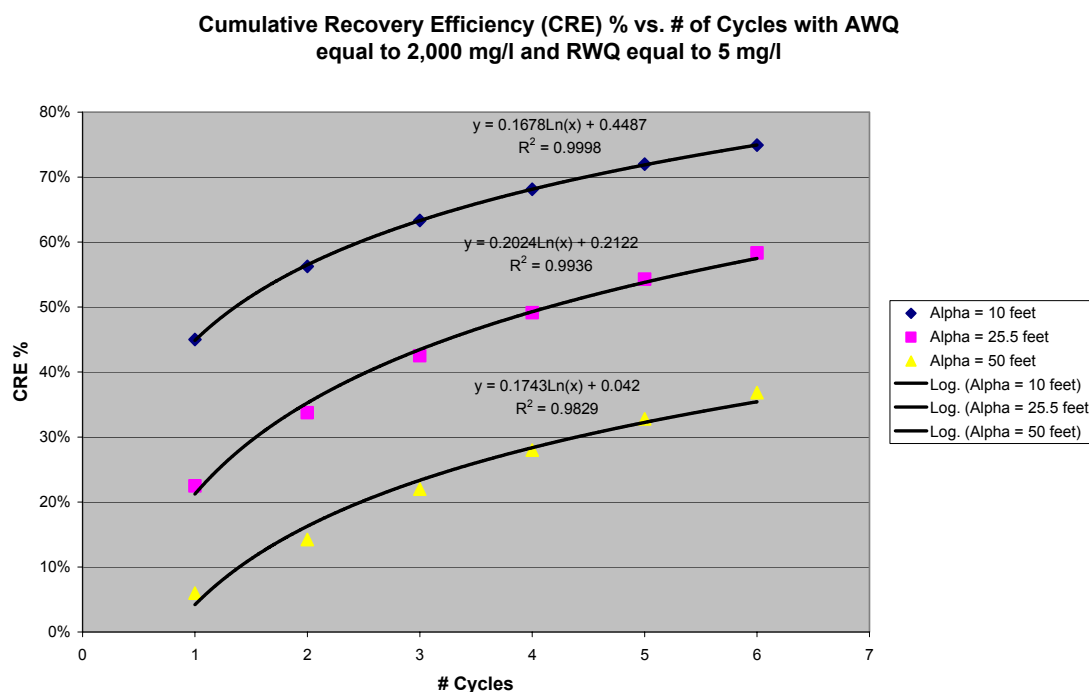


Figure 2-40. Comparison of ASR simulator C predicted CRE% results using ambient groundwater chloride equal to 2,000 mg/l and recharge chloride equal to 5 mg/l, dispersivity is assigned as shown, with logarithm best-fit curves shown

Once the modifications of the transmissivity distribution were completed, the same series of six recharge and recovery cycles was simulated. The results are displayed on Figure 2-41. The model simulations demonstrate that a heterogeneous aquifer would result in slightly worse ASR RE performance than a comparable homogeneous aquifer. In this case, the difference is a 5% reduction in RE percentage for the first three ASR cycles followed by a smaller difference as the number of cycles increase. These data suggest that over time, the reduction in system performance seen in the heterogeneous case would dissipate and eventually no obvious performance difference would be apparent between the two cases. This is a significant conclusion and may indicate that over a long period of time the performance of ASR systems, as measured by RE, may not be that sensitive to the transmissivity distribution. Further modeling simulations may be useful to explore this issue in more detail.

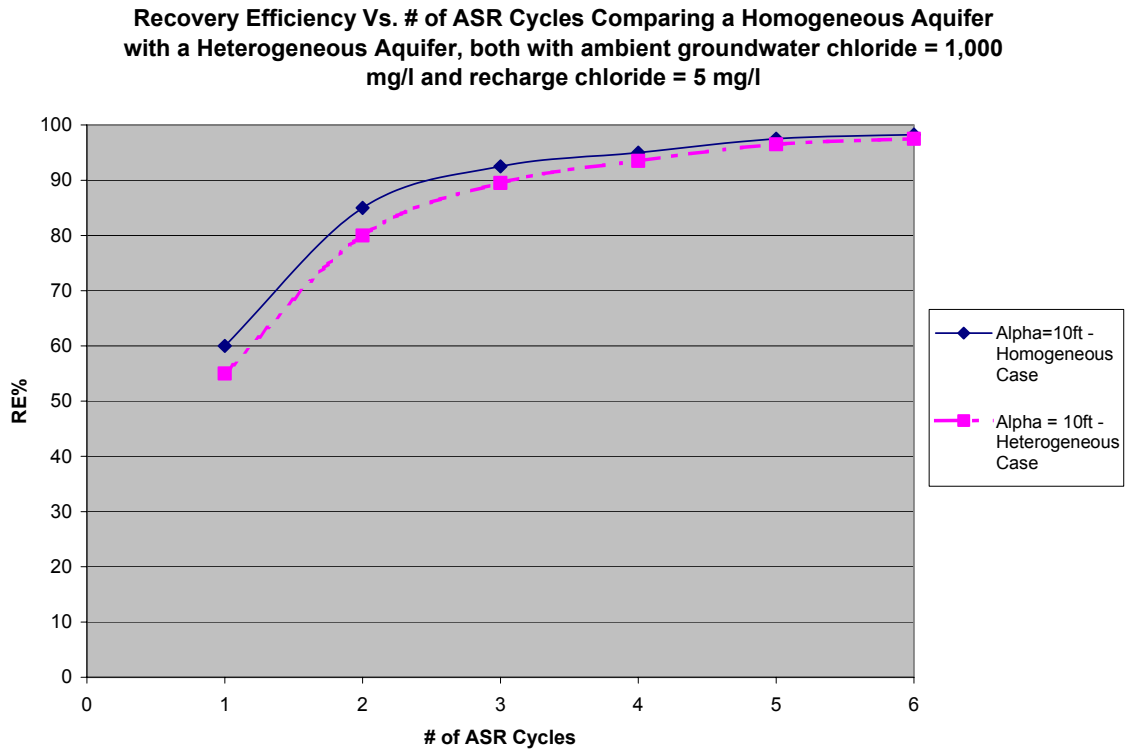


Figure 2-41. Comparison of ASR Simulator C predicted RE% results using ambient groundwater chloride equal to 2,000 mg/l and recharge chloride equal to 5 mg/l, dispersivity is assigned as shown, using both homogeneous and heterogeneous transmissivity distributions

CHAPTER 3

RESULTS AND DISCUSSION – NEW ASR PLANNING DECISION FRAMEWORK

According to Grigg (1996), a coordinated framework “provides a structure for the players to work under, a set of process rules, and a reporting procedure”. Grigg goes on to describe additional attributes widely recognized as essential for management actions based upon decision frameworks including the use of effective tools; the basing of actions on scientific and risk assessment; the use of measures to reduce uncertainty; and the selection of financially feasible approaches. Therefore, a new ASR planning decision framework must provide three functions to likely users. First, the framework should provide the user a general procedure for planning new ASR projects. The procedures can be posed as various questions or steps that need to be undertaken for a typical project. Second, the framework should provide the user guidelines and metrics for ranking and comparison purposes. The guidelines and metrics should aid the user in determining likely project feasibility and cost effectiveness. Third, the framework should direct the user to perform risk and uncertainty evaluations in order to assess the project under multiple future conditions.

Development of ASR Performance Guidelines and Metrics

Performance guidelines and quantitative metrics are important elements of a planning decision framework. Guidelines are usually developed based upon a combination of empirical data, practical experience, and professional judgment. Quantitative metrics are developed through scientific research, modeling, and mathematics. For the new ASR planning decision framework, both guidelines and

metrics have been developed and are discussed in the following sections. Some of the guidelines have been developed for other uses but have applicability to ASR projects, while some of the guidelines were developed from this research effort. The proposed performance metrics were developed in a similar manner.

General Guidelines for all ASR Sites with Consideration of ASR Use for Potable, Irrigation or In-stream purposes

General ASR use and performance guidelines are presented in this section of the report. In each section, consideration is also given to the ASR use as a potable water project, as an irrigation project, or for in-stream environmental purposes. For a number of the guidance categories, the use is immaterial and therefore not emphasized, while for others, guidance may differ depending upon the intended use of the ASR project.

Hydrogeology

The hydrogeology of the proposed ASR storage zone is a key consideration in ASR planning. As was discussed in Chapter 1 of this research report, the existence of a suitable storage zone is a prime constraint to ASR development. What is a suitable storage zone ? Based upon a review of existing ASR sites throughout the United States and the world, a number of attributes are required to be considered a suitable storage zone. The project summaries and comparisons presented in Chapter 2 of this report divulge that both unconfined and confined aquifers can serve as suitable storage zones. For unconfined aquifers, a suitable storage zone should be isolated from sources of anthropogenic contamination as well as groundwater users that could be impacted by the ASR project. The storage zone should contain enough volume to provide the intended water storage. Also, a storage zone “setback” should be included when calculating the required storage volume. The setback provides a buffer area to account for

hydrogeologic irregularities as well as a sentinel monitoring zone. An ideal unconfined aquifer storage zone may have a substantial thickness of unsaturated zone that could serve as the ASR storage reservoir as was the case at the Windhoek, Namibia ASR project (Murray, 2004); the large unsaturated zone could be a resultant of aquifer over pumping, as was the case at the Las Vegas ASR project (Katzner and Brothers, 1989). In addition, a suitable storage zone in an unconfined aquifer should have a moderate to high porosity so that the stored water stays within the vicinity of the ASR well. Aquifers with low porosity tend to be fractured rock aquifers where dual porosity behavior can negatively impact project feasibility (Gale, 2002). A reasonable planning guideline for ASR projects located in unconfined aquifers would be to utilize a minimum thickness of 25 feet with a setback buffer of $\frac{1}{4}$ mile. Figure 3-1 provides a planning nomograph that

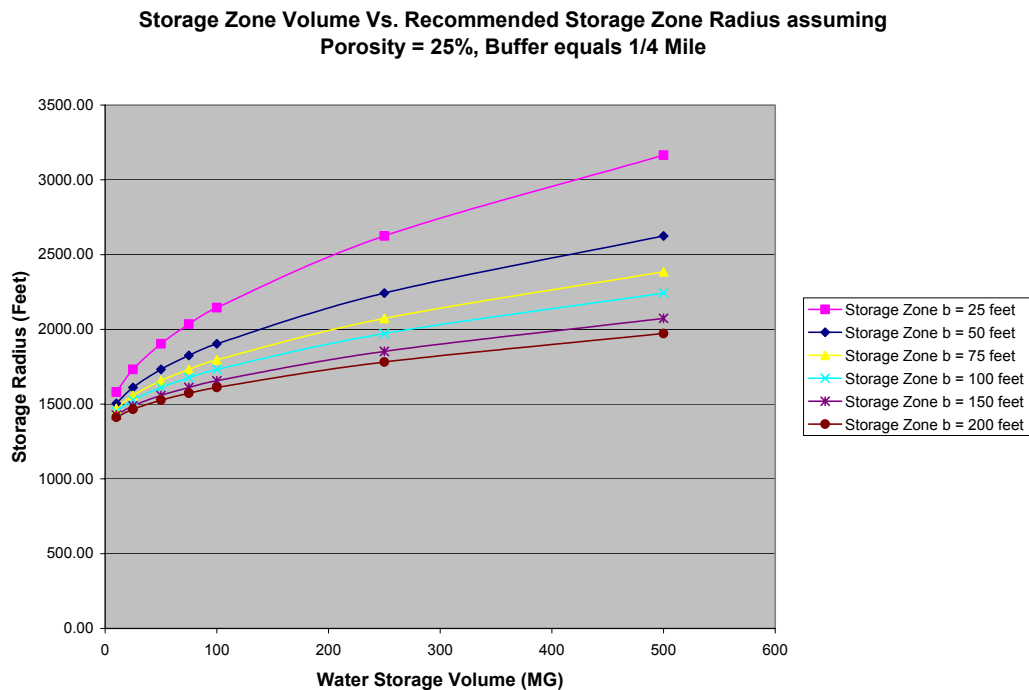


Figure 3-1. Unconfined aquifer ASR planning guideline for water storage volume versus recommended storage zone radius at various aquifer thicknesses (b)

displays the required ASR water storage volume in million gallons (MG) versus the surface radius in feet required with a ¼ mile setback. Also, the figure depicts different ASR storage zone thickness. The recommended storage zone radius should be utilized during the site selection process to ensure that no existing groundwater users or contamination sources are located within the radius from the ASR well. Based upon a review of the existing literature and site summaries developed for this research report, the aquifer storage zone should have a minimum transmissivity of 1,000 ft²/day. The proposed recharge rate should also be hydraulically compatible with the transmissivity (Missimer et al. 2002). Hydraulic guidance values are discussed below.

A majority of ASR projects are completed within confined aquifers instead of unconfined aquifers. This is not surprising given the fact that surface spreading methods are generally more efficient for AR than wells for unconfined aquifers (Abe, 1986; ASCE, 1994). Also, confined aquifers are naturally isolated from the surface by low permeability units. The State of Washington includes “aquifer vulnerability and hydraulic continuity” as one of its standards of review for any new ASR proposal (McChesney, 2001). For confined aquifers, the aquifer transmissivity should be greater than 1,000 ft²/day. The proposed recharge rate should also be hydraulically compatible with the transmissivity. Hydraulic guidance values are discussed below.

The predominant geologic type of the storage zone may also be an important consideration. Although, this report demonstrates that successful ASR projects are located in a variety of different geologic environments, certain types may be subject to higher hydrodynamic dispersion or diffusion. This line of reasoning is discussed further in later sections of this report.

Hydraulic

Hydraulic considerations for any ASR project are of paramount importance. System performance and operability depend upon the ASR practitioners having a clear understanding of hydraulic constraints and effects on efficiency. Hydraulics control the amount of water that a particular ASR project may recharge or recover as well as where and how fast the stored water moves away from the ASR well. Bear and Jacobs (1965) presented the basic mathematics governing the movement of injected water into an aquifer. Bear and Jacobs (1965) illustrated that hydraulics of any injection well system are a function of the aquifer transmissivity, the aquifer gradient, the aquifer geometry, and the overall ASR well stress imparted upon the aquifer.

Planning guidance charts relating the various dependent variables to total head were developed for a series of theoretical ASR projects. Both unconfined and confined aquifers were evaluated using existing analytical model solutions. For the unconfined ASR projects, an analytical curve matching solution developed by Neuman (1975) can be utilized to evaluate anticipated aquifer head increases due to ASR recharge operations. For the confined ASR projects, an analytical solution developed by Theis (1935) was utilized to prepare new ASR hydraulic planning aids. Figures 3-2 to 3-4 are guidance charts that depict predicted head increase versus ASR well recharge rate in MGD. A family of curves representing different aquifer transmissivities is presented on each figure. The three figures display the predicted head increases in a confined aquifer with a storage coefficient of 1.91×10^{-4} and a recharge rate of 1 to 5 MGD for 50 days. The storage coefficient chosen was the geometric mean value determined from a review of 20 non-brackish water ASR projects performed for this research report. The recharge duration of 50 days was chosen to represent a realistic operational scenario. The figures

also show the anticipated head increase at different radii from the ASR well to aid practitioners in ASR cluster design. Using knowledge of superposition in combination with these guidance charts would allow any user to estimate ASR operating heads at a cluster site. The allowable head increase for any ASR project is also constrained due to practical considerations discussed in the following sections.

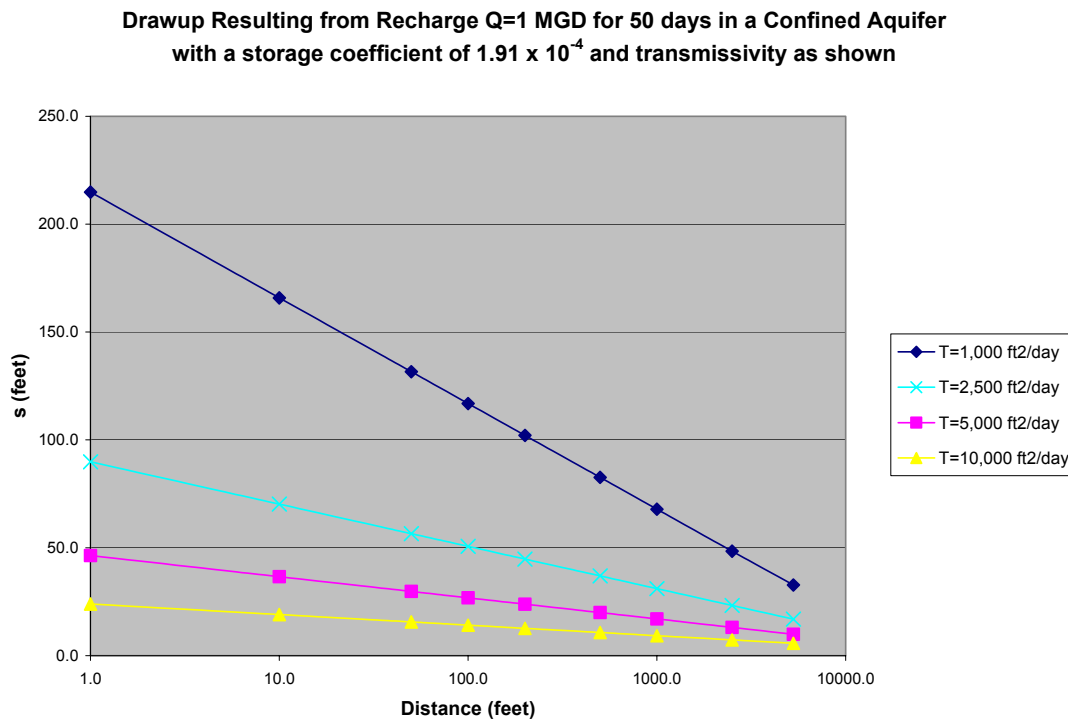


Figure 3-2. Confined aquifer ASR planning guideline at various aquifer transmissivities with recharge rate equal to 1 MGD for 50 days

During ASR recharge or recovery, potentiometric head changes will likely occur. During recharge operations potentiometric heads in the FAS will increase. During recovery operations the converse will be true. The range of expected heads may place some practical engineering constraints upon the system design. Anticipated head increases due to ASR operations should not increase heads in a wide region much beyond the highest observed heads within the aquifer in the study area region. In addition, head

**Drawup Resulting from Recharge $Q=2.5$ MGD for 50 days in a Confined Aquifer
with a storage coefficient of 1.91×10^{-4} and transmissivity as shown**

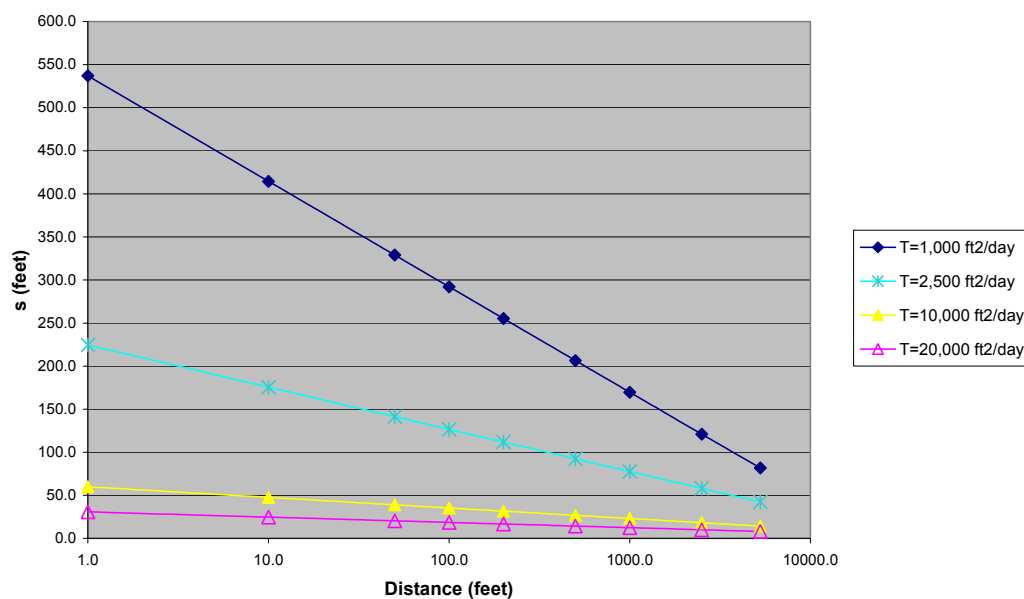


Figure 3-3. Confined aquifer ASR planning guideline at various aquifer transmissivities with recharge rate equal to 2.5 MGD for 50 days

**Drawup Resulting from Recharge $Q=5$ MGD for 50 days in a Confined Aquifer
with a storage coefficient of 1.91×10^{-4} and transmissivity as shown**

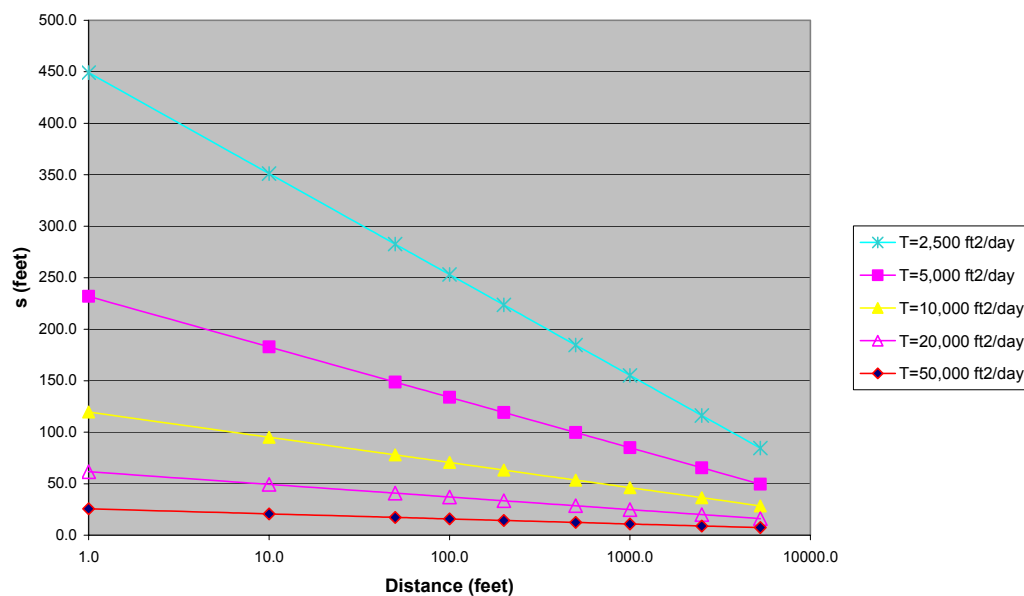


Figure 3-4. Confined aquifer ASR planning guideline at various aquifer transmissivities with recharge rate equal to 5 MGD for 50 days

changes should be managed to avoid the need to install specialized well pumps or motors.

Lastly, the pump motor electricity costs should be constrained to the extent practical.

The expected range of heads from an ASR well field may limit the pump selection for the wells. In addition, the desired maximum ASR well flow rate may limit the available pump selection. For example, at a desired maximum ASR well flow rate of 5 MGD (3,472 gallons per minute) the available pump selection for “off the shelf” models would be quite limited (Brown et al., 2004). Based upon the review of readily available commercial well pumps recommended for deep well applications, the following pump types could be utilized:

- Submersible pumps
- Lineshaft pumps
- Vertical turbine pumps

If submersible pumps are considered, custom design will be required at higher flow capacities since most readily available models are limited to flow capacities of approximately 2,500 gpm. Specially engineered submersible pumps can also be constructed but at significantly higher capital cost. Even with custom designed pumps, pump diameters are limited to 20 inches or less. Also, submersible pumps of this size would require specially manufactured motors that would result in even higher costs. In addition, submersible pumps are usually only used during recovery operations, not recharge operations. Mechanical changes would have to be made to the submersible pumps to allow two-way usage.

If a large operating head range is expected, only larger more specialized lineshaft centrifugal or vertical turbine pumps are applicable. A review of products listed on various pump manufacturer web sites indicates that head ranges of up to 2,500 feet can be

specially engineered. Information from Gould Pumps (<http://www.goulds.com/>) indicates that readily available (off the shelf) one-stage pumps are limited to approximately 200 feet of total dynamic head per stage with a pumping rate of 3,500 gallons per minute (gpm). As heads exceed 200 feet and well pressures approach 100 psi, transient surge pressures and water hammer issues greatly complicate pump selection and force main design. At 100 psi average operating pressures, transient surge pressures could approach 150 to 200 psi. Pressures that high may necessitate selection of specialized high-pressure piping. This type of pipe is much more expensive than comparable steel or PVC piping. One possible option to be considered would be customized pump arrangements with multiple stages.

As the aquifer head increases due to the operation of a well cluster, the individual well pump motor requirements and energy utilization may become quite large. Consequently, electricity demand and costs may provide a practical limitation to the allowable head change during aquifer recharge or recovery. A desktop analysis of theoretical power requirements for individual ASR wells revealed that well pump electricity costs could be considerable as aquifer heads rise. The applicable equation utilized to calculate the Power requirements is (Baumeister et al. 1976) presented as equation (2):

$$\text{Equation (2)} \quad P_w = (g * Q * H) / (3960 * \epsilon)$$

Where P_w is the power in british horse-power (bhp); g is the specific gravity (assumed to be 1.0) of the liquid pumped; Q is the pump flow in gallons per minute (gpm); H is the total dynamic head (feet) to be overcome; and ϵ is the assumed pump hydraulic efficiency.

In addition to ensuring recharge heads do not become excessive, heads during recovery are also important. In confined aquifers, the head changes occur across a very large spatial area. In the best scenario, no existing well users should be located within a one mile radius of the proposed ASR project. If a one mile set back distance is not feasible, a well inventory should be completed to identify existing well pump settings and operational schedules. By collecting this information, the proposed ASR project design can minimize potential impacts to the users such as lowering the piezometric head to below existing pump settings or causing large increases in pump electrical demand.

Besides recharge and recovery heads, another hydraulic performance issue is the pre-existing groundwater aquifer gradient. Bear and Jacobs (1965) present a graphical method of evaluating the hydraulic effects in this case. As was demonstrated by Anderson and Lowry (2004) and confirmed in this chapter of this research report, when the gradient exceeds 0.0005, recovery of stored water is reduced. The magnitude of the reduction is dependent upon the amount of mixing that occurs in the aquifer. At higher dispersivity values, the performance reduction is less pronounced, since hydrodynamic dispersion is a more important performance variable. Based upon the modeling performed in this report, an ideal ASR site would have a gradient less than 0.001. Prospective ASR sites with gradients larger than this may not be good candidate ASR sites, especially if the ambient groundwater quality is poor. If the quality of the source water and groundwater are similar, the pre-existing gradient is less important since recovery of a larger percentage of ambient groundwater would not impact project feasibility.

Geotechnical

ASR well fields are typically designed for the short-term storage of excess water for use during peak demands. In addition, the well fields are designed to recover the maximum amount of water possible at the required demand rate. Many ASR well field storage zones are located in a brackish portion of an aquifer and as a result individual ASR wells tend to be closely spaced to optimize the capture zones and maximize recovery of injected water of superior quality (Brown et al. 2004). Mixing of waters occurs due to a number of mechanisms including groundwater advection, dispersion, and diffusion. Excessive mixing of the injected water with the ambient groundwater can degrade the recovered water quality over time and this can be detrimental to the overall recovery efficiency. This approach, of closely spaced ASR wells, seems reasonable as long as care is taken to evaluate aquifer pressure increases or decreases. ASR wells located in close proximity can lead to significant hydraulic interference with large head increases or decreases as a result. Large well fields such as those proposed by the CERP (USACE and SFWMD, 1999), could lead to extreme head increases or decreases across large areas. It is important to manage these pressure head changes so that significant regional changes are kept to a minimum. Ultimately, an optimization analysis should be performed to minimize aquifer pressure changes while maximizing recovery of injected fresh water. Where no significant difference in water quality occurs, or where the intended use of the recovered water is such that any mixing is acceptable, then more conventional well field design procedures relating to spacing and arrangement of wells may be applicable (Pyne, 1995).

Increased or decreased aquifer pressures can also have unintended geotechnical consequences that should be evaluated carefully. Excess pressures generated during ASR

recharge cycles can potentially lead to hydraulic fracturing of the aquifer medium. In the case of rock aquifers, hydraulic fracturing of the rock itself can greatly increase local-scale aquifer conductivity (National Research Council, 1996). Therefore, injection pressures need to be monitored carefully to ensure that critical pressures are never exceeded. Abnormally high fluid pressures have lead to rock fracture (Domenico and Schwartz, 1998).

Multiple failure mechanisms related to hydraulic fracturing have been identified. First, regional scale shear failure of the rock matrix requires investigation. The stresses developed under this scenario can be evaluated using classical soil mechanics principles such as the “Mohr-Coulomb failure envelope” (Blyth, and de Freitas, 1984). In this type of evaluation, the total vertical and horizontal effective stresses are computed and then combined to estimate the critical normal stress and shear stress that occurs on a failure plane. The second type of failure is the hydraulic failure of the rock matrix. This is thought to occur when the fluid pressure exceeds the least principal stress. Another type of failure mechanism occurs as the pore volume increases due to the formation of microfractures. As the rock matrix dilates due to the increased fluid pressures, microscopic fractures may form. These microfractures may also have preferential directions due to regional stress fields. Handin (1963) reported that dilatancy onset occurs when the ratio of the fluid pressure to the confining pressure is equal to approximately 0.8.

Classical soil mechanics govern the stresses and strains of the rock matrix mass. The governing equation of the overall effective stress state (T) (Terzaghi and Peck, 1967) is expressed as equation (3):

$$\text{Equation (3)} \quad T = C + (\sigma_n - P) \cdot \tan(\Phi)$$

where T is the maximum shear stress, C is the cohesion of the rock determined from triaxial strength testing, σ_n is the normal stress on the projected failure plane, P is the aquifer water pressure, and Φ is the angle of internal friction of the limestone rock.

Using published values (Blyth and de Freitas, 1984) for these terms and an appropriate factor of safety of 1.5, the maximum allowable shear stresses can be developed. In the case of the limestone within the Floridan Aquifer System, a cohesion of 750 psi was selected to represent a weathered limestone; an angle of internal friction of 35 degrees was selected for limestone; and the normal stress was calculated based upon the incremental loads expected from the Everglades project ASR operation (Brown et al. 2004). In general, the cohesion of limestone is the overwhelming source of the resistance to failure or fracturing of the matrix. Therefore, the calculations are very sensitive to the cohesion assumed.

If matrix fracture is a concern to ASR proponents, triaxial strength testing is recommended in order to develop more reasonable estimates of in-situ cohesion. Any testing program should ensure that samples are tested at the proper confining pressures. Wright et al. (2002) recommended a similar need for permeability testing. Values decreased with increasing effective stress due to closure of microfractures caused by sampling for all samples. The study demonstrates the importance of replicating the in-situ effective stresses when measuring hydraulic conductivity of cores of deep aquifers in the lab. In general, matrix fracture would not be initiated until aquifer pressures exceed 600 to 800 psi. For most ASR projects to be feasible, well head pressures would need to be much less than 600 psi so this issue is probably not important for most projects.

The second possible failure mechanism that was investigated was potential hydraulic fracturing. The petroleum industry routinely uses the concept of hydraulic fracturing to enhance the permeability of a rock formation around an oil recovery well. The resulting increase in fractures results in better recovery of petroleum or natural gas. Consequently, much research on this topic (related to the oil industry) is already available in the literature. Hubbert (1957) evaluated the “mechanics of hydraulic fracturing” in a 1957 article in Volume 210 of the AIME Petroleum Transactions. Within this article, Hubbert discusses critical injection stresses necessary to create new matrix fractures and extend existing fractures. He also notes that new fractures would propagate perpendicular to the least principal stress. Basically, the pressure increase required to hold open and extend an existing fracture should be equal to or greater than the least principal stress. In areas of normal faulting, the least principal stress is normally horizontal while in areas of high tectonic activity, the least principal stress may be oriented on the vertical (Hubbert, 1957). Therefore, in tectonically quiescent areas of the United States (such as Florida), the fractures caused by excess pore-water pressure would likely be in the near-vertical orientation. Further exploration of these technical issues in regards to the CERP ASR project is available below.

Hubbert (1957) developed the following relationship for the water pressure required to initiate new fractures or enlarge existing ones. The relationship is shown here as equation (4):

$$\text{Equation (4)} \quad P_f = \sigma_v * M - P^*(M-1)$$

where P_f is the critical stress level to cause new fracture onset or expanded fractures; σ_v is the existing total vertical stress, M is the ratio of horizontal to vertical stress (same as

the lateral earth pressure coefficient at rest, K_0), and P is the ambient (pre-fracture) water pressure. This relationship was utilized to evaluate potential for hydraulic fracturing at the various ASR sites for the Everglades ASR project (Brown et al. 2004). The evaluations clearly show that if total aquifer heads are limited to 250 feet or less, hydraulic fracture initiation is unlikely. The actual fracturing threshold will vary from project to project, however, a total head limit of less than 250 feet seems prudent.

Geotechnical evaluations were completed at the Salem, Oregon ASR project to assess the possibility of vertical hydraulic “jacking” or hydrofracturing. The site consultant recommended that the allowable head increase be limited to 200 feet of buildup (head increase) to minimize jacking effects (Golder Associates, 1996b). Due to well clogging and low aquifer transmissivity, aquifer heads rose 410 feet at the Highlands Ranch ASR project, resulting in probable hydraulic fracturing (Pyne, 1995).

The third evaluation of potential rock fracturing limitations related to ASR projects reviews dilatancy potential. All materials dilate (or change in volume) in response to shearing strains (Domenico and Schwartz, 1998). As the rock matrix dilates due to the increased fluid pressures, microscopic fractures may form. Pore volume increases can lead to the formation of microfractures that may increase the local-scale hydraulic conductivity of the aquifer (Brown et al. 2004). The onset of dilatancy can occur at one-third of the allowable shear stress for a rock matrix. Handin (1963) noted that in sedimentary rocks (including limestone), the ratio of the fluid pressure to the confining pressure should not exceed 0.8. Equation (5) presents the relevant calculation:

$$\text{Equation (5)} \quad \text{Handin Ratio} = P/\sigma_h$$

where P is the aquifer water pressure and σ_h is the least principal stress. Since the least principal stress and the water pressure change as the water pressure increases, the Handin Ratio will also change. For the Everglades ASR project, the maximum allowable head north of Lake Okeechobee as determined from this approach was 183 feet (Brown et al. 2004). Since this is an empirical relationship derived via laboratory testing, the maximum allowable ratio may be somewhat conservative, however, it does appear to provide a useful guide to constrain the anticipated ASR fluid pressures. In fact, the Handin Ratio appears to provide an excellent screening tool for ASR projects concerned with high aquifer pressures within confined bedrock aquifers.

Besides issues related to stress changes within the aquifer, two other geotechnical issues may be important for ASR planning efforts. Hydraulic fracturing of confining units overlying confined ASR storage zones is an issue that may be important and subsidence or rebound of the confining unit itself due to changes in effective stress may constrain ASR development.

The possibility of “hydraulic fracturing” of the Hawthorn Group clay has been discussed by Brown et al. (2004). The literature, however, is clear that fracturing in clays can occur under natural conditions (Hanor, 1993; Gudmundsson et al. 2001) or as a result of construction activities. One ASR project in Thailand reported hydraulic fracturing of a confining unit during pilot testing (Dillon and Pavelic, 1996). In general, the length of “hydrofractures” is short and is generally arrested by a geologic contact where high Young’s modulus materials (hard rock units) intersect with low Young’s modulus materials (clays or marls that compose confining units). Gudmundsson et al. (2001), clearly demonstrates that during extension events propagation of vertical hydrofractures

would be suppressed by soft, pliable layers of marl or clay. In addition, for fracture propagation to continue to the surface, the stress field along its pathway must be homogenous, which is rarely the case.

Land subsidence (downward movement) can be attributed to several causes: tectonic movement; chemical dissolution; consolidation of sedimentary materials; vibrations and water table decline due to groundwater withdrawals. Groundwater withdrawals, however, are one of the most obvious causes for land subsidence (Brown et al. 2004). Land subsidence can be severe where pumping exceeds the aquifer safe yield, and water level or the piezometric surface has declined. Numerous cases of subsidence attributed to groundwater level lowering have been documented. In Baton Rouge, Louisiana, one foot of subsidence has been observed in an industrial district where the piezometric surfaces have dropped about 135 feet due to pumping of groundwater since 1890 (Davis and Rollo, 1969). In San Joaquin Valley, California, pumping of groundwater for irrigation has caused a maximum subsidence of about 32 feet and a total affected area of 5,200 acres where subsidence exceeded one foot (Rutqvist and Stephansson, 2003). Subsidence of clay confining units caused radial cracks to form around ASR wells at the Washoe County ASR project (Bureau of Reclamation, 1996a).

All of the subsidence cases mentioned involved thick clay deposits, which have high compressibility, thus are more vulnerable to subsidence (Brown et al. 2004). The magnitude of the subsidence was directly associated with groundwater level declines that were caused by long-term groundwater extraction. Usually, the greatest magnitude of subsidence occurs where water-level declines are greatest, and where confining units are thicker and most compressible.

The stress history of the deposit also plays an important role in its vulnerability to subsidence (Brown et al. 2004). Generally, soil consolidation is naturally caused by the weight of overlying materials, thus, the stress history depends upon the age and the depth of the deposit, as well as geological processes that the deposit has experienced. The older age and deeper buried the sediment, the greater consolidation it may have experienced. In contrast, a young and shallow deposit may experience less pre-consolidation. Natural phenomena have acted on many soil deposits around the world. Glaciers present within the northern hemisphere subjected local soil deposits to massive loads, erosion, and re-deposition. In Florida, or other marine environments, past sea level stands several hundred feet higher than present day have undoubtedly stressed the existing soil deposits (Miller, 1997). The opposite can also occur. Dissolution of carbonate materials can lead to gradual unloading of a limestone deposit. Therefore, soil deposits have generally experienced a range of historic stresses and loads. Soils subjected to large historic loads (glaciers, sea level rise, etc.) are less likely to subside as compared to virgin deposits of river silts and clays created in recent time. Lambe and Whitman (1969) provide a detailed discussion of consolidation theory as it relates to stress history. Depending upon the soil deposits past stress history, a compressible clay unit could be described as “normally consolidated” or “over consolidated”. Over consolidated clay units have been exposed to very large loads some time in the past that are greater than current loads, whereas normally consolidated units are virgin materials (Brown et al. 2004).

According to Lambe and Whitman (1969), the overconsolidation ratio (or OCR) is a useful metric in determining the susceptibility of a particular deposit to additional

subsidence. The OCR can be determined from one-dimensional consolidation testing of confining unit sediments. The OCR is simply a ratio of the pre-consolidation pressure or stress to the current effective stress. Generally, normally consolidated clays are much more vulnerable to consolidation and consolidate at a faster rate. Normally consolidated clays have OCRs less than 1.0. Over consolidated clays tend to consolidate rather slowly until new stresses exceed those the material may have experienced in the past. Over consolidated deposits have an OCR greater than 1.0. Recent testing as part of the Everglades ASR project indicates that the Hawthorn Group confining unit is highly over consolidated with an OCR of approximately 2.0 (Brown et al. 2004). The OCR can be readily utilized as a guidance value for ASR feasibility studies in order to provide a preliminary check upon potential for subsidence issues.

Water quality

A number of conventional water quality parameters should be evaluated for any ASR project. Total suspended solids (TSS), total organic carbon (TOC), total dissolved solids (TDS), chloride, sodium, and sulfate should all be monitored in both the ASR source water and the recovered water. TSS and TOC values control well clogging including both physical clogging by aggregated particles and clogging due to excessive biological growth on the well screen, in the well filter pack, or in the aquifer itself. Previous studies have shown that the concentrations of TSS and TOC vary widely depending upon the predominant land use type of the basin. Concentrations of TSS were determined to vary from 20 to almost 500 mg/l, while TOC values ranged from 2 to 60 mg/l (Dillon and Pavelic, 1996). Data compiled for this report suggest similar variation. TSS values ranged from less than 1 to 170 mg/l. Smaller variation was observed in TOC values with values ranging from less than 1 to 20 mg/l. The ASCE has published

recommended TSS limits and maximum particle size limits for different well types and different geologic settings (ASCE, 2001). Table 3-1 provides the recommended guidance values from the ASCE.

Table 3-1. Recommended TSS limits from ASCE

Aquifer Type	Recharge Method	Particle Size (microns)	TSS (mg/l)
Alluvial/unconsolidated	Recharge Well	10 to 100	0 to 3
Alluvial/unconsolidated	ASR Well	10 to 100	0 to 5
Karst Limestone	Recharge Well	100 to 500	0 to 5
Karst Limestone	ASR Well	100 to 500	0 to 10
Fractured Bedrock	Recharge Well	100 to 300	0 to 5
Fractured Bedrock	ASR Well	100 to 300	0 to 5

Based upon the literature and the summarized case studies, the recommended value for unconsolidated aquifers seems too high since significant well clogging has been reported at concentrations less than 4 mg/l. A more restrictive value of 0 to 2 mg/l is recommended in this report for use in ASR planning. A lower threshold should be utilized for recharge only wells. Field or laboratory testing may allow ASR designers to recommend higher thresholds than recommended in this report. Table 3-2 provides the final guidance values suggested by this research effort.

TOC values should be kept to a minimum also. TOC have been determined to be the major precursors for the formation of disinfection by-products (EPA, 2001). For this reason, TOC concentrations are recommended to be less than 6 mg/l when chemical disinfection is utilized. In addition, TOC in conjunction with TSS, can interfere with the operation of ultraviolet radiation (UV) disinfection units (Farrell and Burris, 1995) since UV transmittance is a function of light penetration. High TOC values can also lead to excessive biological growth in the well. Excessive biological growths can exacerbate well clogging caused by TSS. Table 3-2 also provides recommended TOC guidance values.

Table 3-2. Recommended TSS and TOC guidelines for ASR projects derived from this research report

Aquifer Type	Recharge Method	TSS (mg/l)	TOC (mg/l)
Alluvial/unconsolidated	Recharge Well	0 to 1	0 to 1
Alluvial/unconsolidated	ASR Well	0 to 2	0 to 2
Karst Limestone	Recharge Well	0 to 5	0 to 4
Karst Limestone	ASR Well	0 to 10	0 to 6
Fractured Bedrock	Recharge Well	0 to 5	0 to 2
Fractured Bedrock	ASR Well	0 to 5	0 to 4

TDS values are regulated in most states and nations. The Federal EPA standard in the United States is 500 mg/l for potable water use. The United Nations recommends a TDS limitation of 450 mg/l to ensure no restriction on irrigation water use and up to 2,000 mg/l with a moderate use restriction for irrigation (Rhoades et al. 1992).

Australian water quality guidelines suggest a maximum limit of 500 mg/l for potable use and 3,000 mg/l for livestock watering (Dillon and Pavelic, 1996). Canada limits TDS values to 500 mg/l for potable water use, 500 to 3,500 mg/l for irrigation use (crop dependent), and 3,000 mg/l for livestock watering (Canadian Water Quality Task Force, 1993). Based upon these references, the recommended TDS guidelines for ASR projects are listed in Table 3-3.

Table 3-3. Recommended TDS guidelines for ASR projects. All values listed in mg/l.

Parameter	Potable Limit	Freshwater Ecosystem Limit	Irrigation Limit	Livestock Watering Limit
TDS	500	500	1,000	3,000

Chloride, sodium and sulfate are also regulated in most states and nations. For chloride, irrigation usage is restricted due to specific plant ion toxicity from high chloride concentrations. The drinking water standard for chloride in the United States is 250 mg/l (EPA, 1998). A similar standard is utilized for potable water in Australia (Dillon and Pavelic, 1996). For environmental in-stream uses of water, the potable standard is also

recommended. For irrigation use, Canada recommends limiting chloride to 100 to 700 mg/l depending upon the crop (Canadian Water Quality Task Force, 1993). The United Nations suggests limiting chloride concentrations to less than 350 mg/l to ensure no use restrictions for irrigation (Rhoades et al. 1992). No standards exist for livestock watering. Based upon these references, the recommended chloride guidelines for ASR projects are listed in Table 3-4.

Table 3-4. Recommended Chloride guidelines for ASR projects. All values listed in mg/l.

Parameter	Potable Limit	Freshwater Ecosystem Limit	Irrigation Limit	Livestock Watering Limit
Chloride	250	250	350	700

For sodium, irrigation usage is restricted due to specific plant ion toxicity from high sodium concentrations. The drinking water standard for sodium in the United States is 250 mg/l (EPA, 1998). A standard value of 180 mg/l is utilized for potable water in Australia (Dillon and Pavelic, 1996). For environmental in-stream uses of water, the potable standard is also recommended. For potable use, Canada recommends limiting sodium to 200 mg/l (Canadian Water Quality Task Force, 1993). The United Nations suggests limiting sodium concentrations to less than 250 mg/l to ensure no use restrictions for irrigation (Rhoades et al. 1992). In addition, the United Nations suggests limiting the Sodium Adsorption Ratio to 3. No standards exist for livestock watering. Based upon these references, the recommended sodium guidelines for ASR projects are listed in Table 3-5.

Table 3-5. Recommended Sodium guidelines for ASR projects. All values listed in mg/l.

Parameter	Potable Limit	Freshwater Ecosystem Limit	Irrigation Limit	Livestock Watering Limit
Sodium	250	250	250	250

For sulfate, irrigation usage is restricted due to specific plant ion toxicity from high sulfate concentrations. The drinking water standard for sulfate in the United States is 250 mg/l (EPA, 1998). A standard value of 500 mg/l is utilized for potable water in Australia (Dillon and Pavelic, 1996). For environmental instream uses of water, the potable standard is also recommended. For potable use, Canada recommends limiting sulfate to 500 mg/l (Canadian Water Quality Task Force, 1993). Limited standards exist for livestock watering. Canada uses a limit of 1,000 mg/l for livestock watering. Based upon these references, the recommended sulfate guidelines for ASR projects are listed in Table 3-6.

Table 3-6. Recommended Sulfate guidelines for ASR projects. All values listed in mg/l.

Parameter	Potable Limit	Freshwater Ecosystem Limit	Irrigation Limit	Livestock Watering Limit
Sodium	250	250	500	1,000

One of the key water quality issues that ASR projects must contend with is disinfection of pathogens. Pathogens were briefly discussed in Chapter 1 of this report and include organisms such as *E. Coli* or *Giardia lamblia* that can sicken or kill potential water users. For ASR projects, they also must be removed or neutralized to comply with applicable drinking water regulations. Removing or neutralizing pathogens has been the subject of intense research over the last decade. For prospective ASR proponents, inactivation of pathogens can be an expensive endeavor.

The survival potential of microbial pathogens in water utilized for ASR recharge is an important consideration for planning and design of ASR projects around the globe. Besides the water supply benefits provided by ASR projects, it has been reported that the quality of the recharged water can be improved through its storage in-situ (Dillon et al.

1999). New research is ongoing to determine if ASR storage can inactivate pathogens through natural attenuation. The potential health risks of recharging stormwater or treated wastewater carrying water-borne pathogens are poorly understood (Toze and Hanna, 2002). Studies have shown that when pathogens are stored in groundwater aquifers, the pathogens exhibit variable rates of decay (Toze and Hanna, 2002). The results of one recent study indicate that bacteria exhibited the largest decay rates followed by coliphage, with enteric viruses decaying the least of all organisms tested (Toze and Hanna, 2002). In a recent study commissioned by the South Florida Water Management District as part of the Everglades ASR project, pathogen inactivation rates (decay rates) were summarized and reviewed (Rose and John, 2003). The study summarized existing literature on the subject and concluded that the only physical property that seemed to correlate to inactivation rates was temperature; the higher the temperature of the groundwater, the larger (faster) the inactivation rate (Rose and John, 2003). In addition, the study concluded that inactivation times for hepatitis A virus were somewhat longer than coliphage, poliovirus 1 and echoviruses (Rose and John, 2003). Lastly, the literature cited in the study seemed also to indicate that inactivation rates for bacteria were larger than viruses and protozoans like cryptosporidium.

From the perspective of a potential ASR practitioner, it appears that the removal of all pathogens from recharge waters, especially viruses and protozoans, is of paramount importance since these pathogenic organisms are quite persistent. In addition, it is apparent that these organisms are ubiquitous in surface waters around the globe (LeChavalier et al. 1991). Unfortunately, current research has not advanced enough to recommend ASR storage durations or storage conditions to provide for pathogen

inactivation. During ASR planning and investigations, an evaluation of indigenous microorganisms in the groundwater storage zone may also be important as research has shown that their presence can increase pathogen inactivation rates (Toze and Hanna, 2002). Currently, ASR projects rely upon conventional disinfection techniques to inactivate pathogens prior to recharge or recovery. Until additional research is performed to demonstrate the beneficial effect of ASR storage upon pathogens, disinfection will continue to be the only alternative for handling pathogens in ASR source water. Therefore, insufficient information exists to provide a firm basis for any new ASR guidance value related to pathogens.

Disinfection by-products or DBPs are a result of various methods of disinfection. DBPs and their formation were briefly introduced in Chapter 1 of this report. In addition, ASR case study reviews presented in Chapter 2 of this report discuss DBPs at a number of project sites. Both non-brackish and brackish water sites can be subjected to issues concerning DBP formation. As was discussed in Chapter 1, the most common form of disinfection is chlorination. Chlorination of source water containing color or moderate concentrations of total organic carbon (TOC) will result in DBP formation. Research completed by the EPA developed the first simplistic conceptual model in the 1970s noting that chlorine plus precursors resulted in THMs (Symons, 1999). Later research discovered that other disinfection techniques also produce various types of DBPs. DBP issues involving ASR projects have been popular research topics over the last 10 years.

In early reconnaissance surveys completed in 1974, THMs resulting from chlorination were discovered to be ubiquitous (Symons, 1999). A frequency distribution developed to determine the relative abundance of the most prominent THMs revealed that

the most abundant substance was chloroform with a median concentration of 21 ug/l as compared to 6 ug/l for bromodichloromethane, 1.2 ug/l for dibromochloromethane, and non-detectable for bromoform (Symons, 1999). Later studies noted that as bromide concentrations in the influent water increase, the resulting concentrations of THMs also increase. After promulgation of the THM rule in 1979,¹ additional utility surveys were completed by the American Water Works Association (McGuire and Meadow, 1988) in order to evaluate what changes resulted from the EPA rule making. The survey determined that the median TTHM concentration among the 1,255 respondents was 38 ug/l. Additional work determined that the second largest class of DBPs was the haloacetic acids (HAAs) and that the ratio of THMs to HAAs was typically 1.45 to 1 (Krasner, 1999).

DBP problems and issues associated with ASR projects have been reviewed by a number of authors over the last 20 years. Roberts et al. (1982) studied the fate of organic chemicals at the Baylands in northern California. Although the work was not definitive, disinfection by-products were noted to attenuate during contaminant transport with chloroform being the most persistent. Both aerobic and anerobic bacteria were mentioned as possibly involved in the attenuation.

Between 1991 and 1993, Singer and Pyne (1993) evaluated the presence of DBPs, specifically total trihalomethanes (TTHMs) and total haloacetic acids (THAAs), in the aquifers underlying five active ASR sites. They found that in all five sites, TTHMs and THAAs were reduced to acceptable levels (TTHMs less than 80 ug/L and THAAs less than 60 ug/L) per regulatory requirements (EPA, 2001) after 75-90 days of storage time.

¹ Federal Register 44 (231):68624-68707

They found that HAA removal occurs rapidly, during aerobic conditions; while THM removal did not occur until anoxic conditions developed. HAA removal precedes THM removal, with more highly brominated species in both classes being eliminated first.

Buszka et al. (1994) studied THM and HAA attenuation at an AR site in Texas. Samples collected 2,900 feet downgradient of the injection point contained key tracers and THMs. Groundwater transport of bromoform and dibromochloromethane was attenuated relative to injected water, chloroform, and dichlorobromomethane. Microbial transformation of bromoform and dibromochloromethane probably was responsible for their apparent disappearance from groundwater. Chloroform and dichlorobromomethane concentrations were mostly affected by advection and dispersion. These compounds were also more persistent than other THMs. The authors conclude relative attenuation of THMs is faster with increased bromination, with chloroform being the most persistent compound. Studies in South Carolina discovered similar phenomena (Mirecki et al. 1998).

In a study in Las Vegas, Landmeyer et al. (2000) examined the potential for induced biodegradation of DBPs within the aquifer at the Las Vegas ASR system site. Through analysis of chloroform and chloroacetic acid, Landmeyer et al. (2000) found a positive potential for biodegradation of chloroacetic acid, but chloroform tended to persist despite the microbial injections. After 100 days, Landmeyer et al. (2000) measured TTHM concentration of less than 50 ug/L and THAA concentration less than 1 ug/L. An ASR test project in Oak Creek, Wisconsin showed that TTHMs degraded to below the state standards in less than six weeks of storage (Miller, 2001). Monitoring of TTHMs revealed that after longer storage times in the aquifer, the various THM

components are degraded anaerobically with dibromochloromethane degrading rapidly and chloroform and bromodichloromethane degrading more slowly.

These previous findings have been corroborated by this research effort through a review of 50 ASR projects. A data review of several non-brackish water ASR sites (Oak Creek, Beaverton, and Hilton Head) strongly indicated biodegradation of DBPs. Generally for these sites, the level of HAAs was noted as declining more rapidly than associated THMs. A few sites reported that of the TTHM monitored or evaluated, chloroform was the most resistant to biodegradation. At the Green Bay site, bromodichloromethane was contained within the recovered water. A data review of several brackish water sites (Charleston and Palm Bay) indicated possible biodegradation of DBPs, although the data are not as definitive as similar sites noted for the non-brackish water projects. Generally for these sites, the level of HAAs was noted as declining more rapidly than associated TTHMs. Data from the ASR site in Tampa also demonstrated that of the TTHM monitored or evaluated, chloroform was the most resistant to biodegradation. The distribution of TTHMs versus HAAs at the 50 ASR sites was also similar to those reported by previous researchers. For instance, average concentrations of TTHMs and THAAs at the Salem, Oregon ASR site were 35 ug/l and 26 ug/l, respectively (Golder Associates, 1996c). Both the THM concentration and the ratio of TTHMs to THAAs were very close to those reported by Krasner et al. (1989). The Las Vegas ASR project had a concentration of TTHM of 50 ug/l, only slightly higher than the median reported by Krasner (1999).

It appears that both water quality mixing and bioremediation play a role in the reduction of DBPs. Likely processes include dispersion, chemical degradation, microbial

degradation and adsorption (Nicholson et al. 2002). Nicholson determined that aerobic and anaerobic bacteria were successful in degrading HAAs, while THMs could only be degraded by anaerobic bacteria (Nicholson et al. 2002). Reportedly, removal of nitrogen compounds including nitrates has also been monitored along with DBP removal (Jones et al., 1999). Denitrification can occur prior to DBP reduction and is not uncommon (Pyne, 1995). Fitzpatrick (1986) reported that at an ASR site in Lee County, Florida, decreases in DOC, organic nitrogen, and nitrate occurred. Another recent research effort focused upon the decay of DBPs under controlled conditions in a simulated sand aquifer (Carlson and McQuarrie, 2003). The study noted that DBPs did degrade over time in the aquifer and that the sequencing of the chlorination process can impact DBP formation (Carlson and McQuarrie, 2003).

Additional research on fate of disinfection by-products during ASR storage has been developed recently (Pavelic et al. 2005b). This research effort is probably the most comprehensive to date on the subject. The new research has provided the first quantitative estimate of THM and HAA half-lives. The results revealed that the half-lives for the THMs varied from 1 to 65 days, while the half-lives for the various HAAs were rapid at less than one day (Pavelic et al. 2005b). Also, chloroform was noted to have the highest persistence while bromoform had the lowest. This recent work is consistent with previous literature and the data reviewed for this research effort. One new finding is that the attenuation rates of THM compounds were shown to be dependent upon redox condition within the aquifer (Pavelic et al. 2005b). Methanogenic conditions seemed to favor rapid biodegradation rates, while nitrate-reducing conditions led to slower rates.

The case study review performed in Chapter 2 also revealed that three ASR sites had issues with the in-situ formation of DBPs. Fram et al. (2003) found that TTHMs continued to form in the aquifer injection zone at the Lancaster ASR site until the residual chlorine present in the injected surface waters was used up. ASR projects in Las Vegas and Boynton Beach had similar problems. Generally, for these three sites, the recharged source water contained moderate TOC concentrations and chlorine residuals of up to 4 mg/l.

EPA has published regulatory guidance for TTHM, HAAs, bromate, chlorite, and the various disinfectants themselves (EPA, 2001). Table 3-7 lists the relevant drinking water limits for potable water plants. Generally, EPA recommends removal of TSS and TOC prior to chemical disinfection since these precursors are responsible for the formation of the DBPs. Table 3-8 lists required removal percentages of TOC.

Table 3-7. Regulatory limits for disinfection by-products and disinfectants themselves

Regulated Contaminants	Maximum Contaminant Level (mg/l)
Total Trihalomethanes (TTHM)	0.080
Chloroform	
Bromodichloromethane	
Dibromochloromethane	
Bromoform	
Five Haloacetic Acids (HAA5)	0.060
Monochloroacetic acid	
Dichloroacetic acid	
Trichloroacetic acid	
Bromoacetic acid	
Dibromoacetic acid	
Bromate (plants that use Ozone)	0.010
Chlorite (plants that use Chlorine Dioxide)	1.000
Chlorine	4.000 as Cl ₂
Chloramines	4.000 as Cl ₂
Chlorine Dioxide	0.800

Table 3-8. Required percentage removal of TOC

Source Water TOC (mg/l)	Alkalinity, 0 to 60 mg/l as CaCO ₃	Alkalinity, 60 to 120 mg/l as CaCO ₃	Alkalinity, 120 mg/l and greater as CaCO ₃
2.0 to 4.0	35%	25%	15%
4.0 to 8.0	45%	35%	25%
> 8.0	50%	40%	30%

*Source, EPA 2001.

While these guidance values are useful for potable water ASR projects and irrigation water projects where crops are intended for human consumption, little guidance is available for ASR sites to be used for in-stream flow purposes or alternate agricultural use such as livestock watering. Canada has developed recommended water quality guidelines for a number of THMs for freshwater ecosystems (Canadian Water Quality Task Force, 1993). Chloroform is limited to 1.8 ug/l, while chloromethane is limited to 98.1 ug/l. The guidance value for chloroform is very low and given the persistence of chloroform revealed in this research report, might represent a serious constraint to ASR usage in in-stream situations.

Most epidemiological studies of DBP exposure have examined the risk to human health from consuming THMs or HAAs (Zavaleta et al. 1999). All chemical disinfectants are known to form various types of DBPs, however, only a handful of the 37 known DBPs are currently regulated (Krasner, 1999). Animal studies have uncovered evidence for carcinogenicity related to five DBPs including dichloroacetic acid, trichloroacetic acid, chloral hydrate, bromate, and MX (Zavaleta et al. 1999). All of these compounds with the exception of chloral hydrate and MX are already regulated. The remaining DBPs have either not been linked to increased cancer risk or they have not been studied in detail yet. In consideration of environmental ASR projects that utilize a chemical disinfectant, concentrations of chloral hydrate and MX should be monitored in

addition to the regulated contaminants since studies have shown that these compounds also have mutagenic effects (Zavaleta et al. 1999).

For agricultural use of the water for livestock watering, Canada has published some guidance criteria for a handful of THMs (Canadian Water Quality Task Force, 1993). The concentrations of bromoform, chloroform, and dichlorobromomethane are limited to 100 ug/l, while the concentration of chloromethane (methyl chloride) is limited to 50 ug/l.

As was discussed in Chapter 2 of this report, ASR performance can also be negatively affected by geochemical reactions that occur between the aquifer matrix and the injected water or the ambient groundwater and the injected water. Gale (2002) noted that geochemical issues were one concern that could constrain ASR development in England. Geochemical issues have been identified at a number of ASR project sites reviewed in this report. According to the EPA (1999b), the intended use or recharge objective needs to be considered in evaluating quality of water required. Generally, environmental receptors will be more sensitive to geochemical contaminants than humans or agricultural crops. Geochemical issues related to ASR projects are technically difficult and sometimes impossible to predict accurately. The ASR practitioner should perform adequate geochemistry studies during the feasibility stage to anticipate and ultimately minimize potential problems to the extent possible. Studies should include an evaluation of the source water and ambient groundwater compatibility, studies to evaluate the aquifer matrix materials, and geochemical desktop modeling to determine problematic compounds. Limited geological explorations should be completed to evaluate the aquifer for geochemical compatibility. It is recommended that rock cores be submitted for

mineralogic testing to assess the amount and type of clay, the amount and form of pyrite and other minerals, the effective porosity, and the vertical and horizontal permeability.

Testing should include the following:

- X-ray diffraction mineralogical analysis (XRD)
- Thin Section Petrography
- Scanning Electron Microscopy (SEM)
- Acid insoluble residue analysis
- Cation exchange capacity and leachate analysis
- Laser or sieve particle analysis
- Helium porosity, air permeability, and grain size density
- Air permeability analysis (vertical plug orientation)
- Energy dispersive x-ray analysis
- Specific gravity analysis
- Core slabbing macroscopic core description and photography

Sections chosen for additional specific analyses should be those that contain visible metallic deposits as well as sections that have a higher porosity that would be representative of the areas in each aquifer where the majority of water would flow (CH2M Hill, 2000b).

The literature review and ASR case studies compiled for this report indicate that a number of heavy metals and halides have been problematic contaminants due to geochemical reactions between the source water, ambient groundwater and the aquifer matrix. Prominent contaminants include aluminum, arsenic, fluoride, cobalt, copper, iron, lead, manganese, mercury, nickel, radioactive substances (e.g., combined radium and uranium), selenium, and zinc. The recommended water quality limits for these

contaminants vary widely depending upon planned water uses. For example, the State of Florida potable limit for aluminum is 100 ug/l, the freshwater ecosystem limit is 5 to 100 ug/l, the marine water ecosystem limit is 1.5 mg/l, and the irrigation or livestock watering limit is 5 mg/l (Florida Administrative Code - FAC, 1996). Obviously, the guidance values vary by three orders of magnitude. For this report, guidance values have been compiled from the State of Florida, EPA, Canada, and Australia/New Zealand. Table 3-9 presents this information for key heavy metals and halides identified as ASR geochemical concerns. For each parameter, the first row is the State of Florida guidance value (FAC, 1996), the second row is the EPA guidance value (EPA, 1996), the third row is the Canadian guidance value (Canadian Water Quality Task Force, 1993), and the last row is the Australian/New Zealand guidance value (Dillon and Pavelic, 1996; ANZECC, 2000).

Table 3-9. Regulatory Guidance Values from the State of Florida, EPA, Canada, and Australia/New Zealand. All concentrations listed in ug/l except radium.

Parameter	Potable Use	Fresh Water Ecosystem	Marine Water Ecosystem	Irrigation or Agricultural Supply	Livestock Watering
Aluminum	100 50 ***	---- ---- 5 to 100 # ----	1,500 ---- ---- ----	---- ---- 5,000 5,000	---- ---- 5,000 5,000
Antimony	14 6 6 ----	4,300 ---- ---- ----	4,300 ---- ---- ----	---- ---- ---- ----	---- ---- ---- ----
Arsenic – Total	50 10 25 7	50 340 5 ----	50 69 12.5 ----	50 ---- 100 100	---- ---- 25 500
Arsenic – Trivalent	---- ---- ---- ----	---- ---- ---- 24	36 ---- ---- ----	---- ---- ---- ----	---- ---- ---- ----

Notes: * - Average annual concentration; # - Hardness dependent minimum value; ** - under review; *** - Secondary drinking water standard

Table 3-9. Continued.

Parameter	Potable Use	Fresh Water Ecosystem	Marine Water Ecosystem	Irrigation or Agricultural Supply	Livestock Watering
Barium	1,000 2,000 1,000 ----	---- ---- ---- ----	---- ---- ---- ----	---- ---- ---- ----	---- ---- ---- ----
Beryllium	0.0077* 4 ---- ----	0.13* ---- ---- ----	0.13* ---- ---- ----	100 to 500 # ---- 100 ----	---- ---- 100 ----
Boron	---- ---- 5,000 300	---- ---- ---- 370	---- ---- ---- ----	750 ---- 500 to 6,000 500 to 8,000	---- ---- 5,000 5,000
Cadmium	0.37 # 5 5 2	0.37 # 2 0.017 0.20	9.3 40 0.12 5.5	---- ---- 5.1 10	---- ---- 80 10
Chromium – Hexavalent	11 ---- ---- ----	11 16 1 1	50 1,100 1.5 4.4	11 ---- 8 ----	---- ---- 50 ----
Cobalt	---- ---- ---- ----	---- ---- ---- ----	---- ---- ---- 1	---- ---- 50 ----	---- ---- 1,000 ----
Copper	2.85 # 1.30 ** 1,000 1,500	2.85 # 13 2 to 4 # 1.4	3.7 4.8 ---- 1.3	500 ---- 200 # 200	---- ---- 500 # 500
Zinc	37 # 5,000 *** 5,000 3,000	37 # 120 30 8	86 90 ---- 15	1,000 ---- 1,000 # 2,000	---- ---- 50,000 20,000
Combined Radium	5 5	5 ----	5 ----	5 ----	---- ----
Ra 226	1.1 Bq/l	----	----	----	----
Ra 228	----	----	----	----	----

Notes: * - Average annual concentration; # - Hardness dependent minimum value; ** - under review; *** - Secondary drinking water standard

Table 3-9. Continued.

Parameter	Potable Use	Fresh Water Ecosystem	Marine Water Ecosystem	Irrigation or Agricultural Supply	Livestock Watering
Fluoride	1,500 4,000 1,500 ----	10,000 ---- ---- ----	5,000 ---- ---- ----	10,000 ---- 1,000 ----	---- ---- 1,000 ----
Iron	300 300 *** ---- 300	1,000 ---- ---- ----	300 ---- ---- ----	1,000 ---- 5,000 1,000	---- ---- ---- ----
Lead	0.55 # 15 ** 10 10	0.55 # 65 1 to 7 # 3.4	8.5 210 ---- 4.4	50 ---- 200 200	---- ---- 100 100
Manganese	---- 50 *** 50 500	---- ---- ---- 1.9	---- ---- ---- ----	---- ---- 200 2,000	---- ---- ---- ----
Mercury	0.012 2 1 ----	0.012 1.4 0.026 0.6	0.025 1.8 0.016 0.4	0.2 ---- ---- ----	---- ---- 3 ----
Nickel	1 # ---- ---- ----	1 # 470 25 to 150 # 11	8.3 74 ---- 70	100 ---- 200 ----	---- ---- 1,000 ----
Molybdenum	---- ---- ---- ----	---- ---- 73 ----	---- ---- ---- ----	---- ---- 10 to 50 # ----	---- ---- 500 ----
Selenium	5 50 10 ----	5 5 ** 1 11	71 290 ---- ----	---- ---- 20 to 50 # ----	---- ---- 50 ----
Silver	0.07 ---- ---- ----	0.07 3.2 0.10 0.05	---- 1.9 ---- 1.4	---- ---- ---- ----	---- ---- ---- ----
Vanadium	---- ---- ---- ----	---- ---- ---- ----	---- ---- ---- 100	---- ---- 100 ----	---- ---- 100 ----

Notes: * - Average annual concentration; # - Hardness dependent minimum value; ** - under review; *** - Secondary drinking water standard

The guidance values recommended should be utilized in the following manner. First, the appropriate State or Federal criteria should be reviewed by the ASR practitioner. If no guidance values have been promulgated, the user should then consult this compilation table to determine if another entity has published recommended water quality limits. In the category of livestock watering, limited guidance has been found in the literature. If no values are available, the user should consult any epidemiological information that has been published in the literature.

In general, toxics include a broad range of contaminants found in storm water including volatile organics, semi-volatile organics, pesticides, herbicides, and algal toxins such as microcystin. Toxics are already highly regulated due to identified human health issues (EPA, 1998). As discussed in Chapter 1 of this research report, many toxics are regulated under the Safe Drinking Water Act and as such, cannot be present in recharge waters in concentrations greater than required limits. Therefore, for potable water ASR projects, toxics must be removed via water treatment prior to ASR recharge or the source water must not contain these compounds at all. A similar requirement is necessary for ASR projects planned for use to irrigate crops grown for human consumption. ASR projects supporting in-stream water use will need to pay particular attention to toxics since only a limited number of water quality standards have been promulgated related to toxics released into the environment. Obviously, for ASR in-stream projects, the source water is also the receiving water. Therefore, ASR projects are not likely to exacerbate any water quality problems, however, avoidance of source waters containing toxics is recommended. In the case of in-stream water supply projects, pesticides, herbicides, and algal toxins should be evaluated since these compounds have been identified in many

surface water bodies (EPA, 1999c). Also, water quality criteria for these substances can be more restrictive in the case of environmental receptors. For this report, guidance values for selected compounds have been compiled from the State of Florida, EPA, Canada, and Australia/New Zealand. Microcystin is also listed based upon guidance from the World Health Organization (WHO, 1998). Table 3-10 presents this information for key pesticides, herbicides, and microcystin. For each parameter, the first row is the State of Florida guidance value (FAC, 1996), the second row is the EPA guidance value (EPA, 1996), the third row is the Canadian guidance value (Canadian Water Quality Task Force, 1993), and the last row is the Australian/New Zealand guidance value (Dillon and

Table 3-10. Regulatory Guidance Values from the State of Florida, EPA, Canada, and Australia/New Zealand for select pesticides, herbicides, and microcystin. All concentrations listed in ug/l

Parameter	Potable Use	Fresh Water Ecosystem	Marine Water Ecosystem	Irrigation or Agricultural Supply	Livestock Watering
Aldicarb	---- ---- 9 ----	---- ---- 1 ----	---- ---- 0.15 ----	---- ---- 54.9 ----	---- ---- 11 ----
Aldrin	3 ---- 0.70 ----	3 3 ---- ----	1.3 1.3 ---- ----	---- ---- ---- ----	---- ---- ---- ----
Atrazine	---- 3 5 ----	---- ---- 1.8 13	---- ---- ---- ----	---- ---- 10 ----	---- ---- 5 ----
b-BHC	0.014 * .0091 ---- ----	0.046 * ---- ---- ----	0.046 * ---- ---- ----	---- ---- ---- ----	---- ---- ---- ----
Chlordane	0.0043 2 ---- ----	0.0043 2.4 ---- 0.08	0.004 0.09 ---- ----	---- ---- ---- ----	---- ---- ---- ----

Notes: * - Average annual concentration; # - Value from the WHO, 1998;

Table 3-10. Continued

Parameter	Potable Use	Fresh Water Ecosystem	Marine Water Ecosystem	Irrigation or Agricultural Supply	Livestock Watering
Chlorpyrifos	----	----	----	----	----
	----	----	----	----	----
	90	0.0035	0.002	----	24
	----	0.01	0.009	----	----
Diquat	----	----	----	----	----
	20	----	----	----	----
	70	----	----	----	----
	----	1.4	----	----	----
DDT	0.001	0.001	0.001	----	----
	----	1.1	0.13	----	----
	----	----	----	----	----
	----	0.01	0.01	----	----
Endosulfan	0.056	0.056	0.0087	----	----
	----	0.22	0.034	----	----
	----	0.02	----	----	----
	----	0.20	0.008	----	----
Endrin	0.0023	0.0023	0.0023	----	----
	2	----	----	----	----
	----	----	----	----	----
	----	0.02	0.008	----	----
Lindane	0.08	0.08	0.16	----	----
	0.20	0.95	0.16	----	----
	----	0.01	----	----	----
	----	0.20	----	----	----
Malathion	0.1	0.10	0.10	----	----
	----	----	----	----	----
	190	----	----	----	----
	----	0.05	----	----	----
Microcystin	1 #	----	----	----	----

Notes: * - Average annual concentration; # - Value from the WHO, 1998;

Pavelic, 1996; ANZECC, 2000). Many of the listed compounds will not be contained in all prospective source waters, however, as was noted in Chapter 1, the cyanotoxin microcystin is likely to exist in 80% of source waters (Carmichael, 2001). Luckily, Carmichael (2001) determined that only 3.4% of water samples had concentrations of microcystin higher than the WHO standard. Since microcystin potential is high in most source waters, it should be added to the source water monitoring program.

As was discussed in Chapter 1 of this research report, the occurrence and fate of pharmaceutically active compounds (PhAC) and personal care products (PPCPs) is a new area of intensive research. Several studies have documented the presence of these contaminants within wastewater plant effluent (Hirsch et al. 1999; Kolpin, 2002). A recent study undertaken by the American Water Works Association (AWWA) generally confirms the previous studies (Sedlak et al. 2005). The AWWA study developed estimates of the occurrence of PhAC compounds within wastewater in the United States. In addition, the study evaluated the limited number of analytical techniques that can be used to detect these compounds during water quality monitoring. The report provides a number of conclusions that may be of interest to ASR practitioners including:

- Acidic drugs, beta-blockers and antibiotics often are present in the effluent of conventional municipal wastewater treatment plants at concentrations between 10 and 10,000 ng/l
- PhACs appear to be removed effectively in advanced wastewater treatment plants equipped with reverse osmosis or granular activated carbon
- Most PhACs appear to be removed during soil aquifer treatment
- Little removal of PhACs occurred in two engineered treatment wetlands studied, even using long residence times of up to one week

The report also discusses the use of PhACs on livestock including poultry, swine, and cattle. Livestock feeding operations could be huge sources of PhACs discharged to the environment since of the 6,600 estimated number of facilities, only 25% have discharge permits required by the Clean Water Act (Sedlak et al. 2005).

These compounds are a concern for all ASR projects, however, they may be especially important for instream projects. As was discussed in Chapter 1, environmental receptors may be more sensitive than humans to low doses of these contaminants or possible synergistic effects of multiple compounds. For ASR in-stream projects, the

source water body is also the discharge body. In one respect, ASR simply changes the distribution and timing of water in the system; therefore, ASR may not exacerbate any potential environmental problems that result from these compounds.

In order to minimize the concentrations of these substances, it is recommended that ASR site selection studies determine where key sources of these contaminants are located. The two main sources identified in this report are wastewater treatment plants and livestock feeding operations. The best option would be to avoid these facilities altogether by siting the ASR project in alternate locations, however, if this is not possible, a setback criterion should be considered. A minimum setback distance downstream of one of the problematic facilities should be one mile. If the ASR project is to be co-located with a wastewater treatment plant or if the wastewater plant is to provide the ASR source water, water treatment options to remove the PPCPs and PhACs is strongly recommended for consideration.

Environmental impacts

Environmental impacts of any ASR project should be minimized to the extent practical. In general, no new water resources project will have zero environmental impacts. Good planning can be utilized to ensure that any such impacts imparted from a new ASR project are negligible. Williams et al. (2001), relates three general outcomes that could be expected to occur during a preliminary review of ASR related environmental impacts including:

- Impacts will be negligible
- Impact is significant so that the site may not be suitable
- Further investigations are required

In a majority of ASR project summaries compiled for this research report, environmental impacts were indeed negligible, however, it is logical to conclude that some projects will have moderate to significant impacts. The Washoe County, Nevada ASR project summarized in Chapter 3 of this report is a good example of this possibility. In order to evaluate potential environmental impacts of an ASR project, impact categories have been derived as part of this research effort. Based upon the available literature and ASR case histories, prospective ASR projects can cause the following categories of environmental impacts:

- Water diversions from environmental users in source water bodies
- Temperature changes in source water bodies during ASR recovery operations
- Water quality changes in source water bodies during ASR recovery operations
- Reduction of water levels in unconfined aquifers during ASR recovery operations
- Increase of water levels in unconfined aquifers during ASR recharge operations

The first major category is water diversions. As was noted in Chapter 1 of this report, source water availability is a major ASR project constraint. Obviously, if water is not available for ASR storage, the project cannot be feasible. The availability of source water should be determined by a thorough inventory of existing water uses and demands, including those required for the source water ecosystem. The Washoe County ASR project was ultimately determined to be infeasible due to the source water body (e.g., the Truckee River) being over allocated already (Bureau of Reclamation, 1996a). System demands are categorized into type and location. Common demand types include:

- Public supply
- Irrigation – Agricultural
- Irrigation – Non-agricultural (parks, golf courses, etc.)
- Non-contact cooling
- Industrial supply
- Ecological

The known demands are inventoried and the remaining demands must be estimated based upon zoning, demographics and population trends. For estimating public demands, 140 gallons per day per capita is often utilized (National Research Council, 2002b). For irrigation or agricultural demand estimates, water use models can be developed. The St. Johns River Water Management District uses the Modified Blaney-Criddle and other similar models to develop irrigation demands (CH₂M Hill, 1997). In addition to known demands, other uses of the source water body that could be impacted by water diversions should be evaluated including navigation and recreation. Once all of the demands and uses are compiled, the excess available water can be determined using a monthly or seasonal water budget. Care must be taken to accurately estimate the real ecological water needs since they have been chronically underestimated in the past (National Research Council, 1997).

Temperature differences between the recovered ASR water and the discharge water body (if any) are an important consideration for ASR projects providing in-stream water supply to environmental users. Potable or irrigation uses of ASR recovered water are less sensitive to large temperature differences. Temperature changes of a few degrees centigrade can negatively affect fishery resources or benthic organisms. Typically, temperature changes are regulated similar to certain contaminants; a mixing zone is used along with monitoring or sampling. For example, an ASR well discharging 5 MGD into a stream with a similar base flow would be a one to one mixing ratio. Therefore, the resultant temperature of the mixed water would be an average of the recovered water and the ambient stream water. Similar to geochemical considerations, temperature compatibility should be considered during ASR system design.

Water quality changes resulting from ASR recovered water discharges have been discussed previously in this report. Water quality guidance values and limits have been compiled and recommended levels specified. In addition to adhering to recommended water quality guidelines, some in-stream projects may need to study the effect of recovered water on existing environmental communities. Horvath (1973) studied the transport of various heavy metals in freshwater and estuarine environments of the Big-Cypress Preserve area of south Florida. Dooris and Martin (1979) documented the changes in basic water chemistry of a freshwater lake augmented with AR from the Floridan Aquifer System. At an AR project in Texas, the treated wastewater was tested for biological toxicity (Bureau of Reclamation, 1996c). Biototoxicity testing of the treatment plant product water was evaluated using bioassays according to EPA (1989). Based upon monitoring and testing, no significant differences existed between control water and the product water. The treated wastewater was concluded to be non-toxic to aquatic organisms. As part of the ASR Everglades study, bench-scale “treated” surface water and native groundwater from the Floridan Aquifer System were utilized in acute and chronic effect toxicity testing (Golder and Associates, 2005). The test results revealed that the treated surface water did not cause any major acute or chronic effects to the species tested including fathead minnows, *Daphnia magna*, *Ceriodaphnia dubia*, and FETAX. Only minor effects (slightly lower reproduction rate) were noted for the 7-day chronic test for *Ceriodaphnia dubia*. Similar reduced reproduction rates were noted using 100% natural groundwater. When the groundwater was diluted 50% with treated surface water, no effect was detected (Golder and Associates, 2005). Based upon the literature reviewed, toxicity testing and bioassays may be prudent measures to undertake

during an ASR feasibility study for ASR projects recommending in-stream use of recovered water.

Changes in water levels in unconfined aquifers have been discussed previously in this report but it is important to re-emphasize water level changes in this section of the report since unconfined aquifers may be intimately connected to natural ecosystems such as freshwater wetlands. Wetland ecosystems may be sensitive to both increased water levels and decreased water levels. Davis (1994) has documented the sensitivity of freshwater wetlands within the Everglades to changes in hydroperiod and water quality. ASR recharge in unconfined aquifers may increase water levels leading to longer hydroperiods in wetland systems. ASR recovery may result in shorter hydroperiods or over drying. River and stream systems that are connected to subsurface aquifers may be similarly affected.

Legal, institutional and social considerations

All ASR projects should consider legal, institutional and social planning factors. The general regulatory framework for ASR projects was described in Chapter 1 of this report. Additional legal and institutional planning considerations include consideration of “water rights” or ownership of the stored water.

The role of “water rights”, especially in the western portion of the United States, is an important planning consideration. In the western states the Prior Appropriation Doctrine governs water rights. This doctrine establishes the right of a water user to use a specific amount of the state’s water resources for a specific beneficial use as stated in the water right by rule or statute (AWWA, 2002). Water rights are established by date with senior and junior rights holders. Senior water rights generally have first access to the appropriated water and on occasion may deny water allocations to more junior rights

holders. Water users are restricted from changing the way they withdraw, divert, or use their water so that those changes do not impact other existing water rights holders (AWWA, 2002). ASR projects may be impacted by water rights issues since they store water and recover water at different times of the year. The Alamogordo ASR project identified three key water-rights questions in connection with the planning of ASR projects:

- Whether storage in the aquifer is a use permitted under the existing right to divert surface water
- Whether recovery of the stored water is permitted under the existing rights to produce groundwater from wells
- How the ASR water might be protected from appropriation by others (Livingston and Finch, 2003)

The third question seems to be an important issue for larger ASR projects. The Palm Bay ASR project noted that a nearby water user diverted stored water from the ASR site through pumping effects. The Everglades Restoration ASR program envisions injecting up to 1.6 billion gallons per day of excess stormwater into the brackish Floridan Aquifer System using up to 303 deep wells. Obviously, a project of this magnitude will need to store water across a large areal expanse in order to minimize any environmental impacts. Approximately 200 of the wells are proposed in an area north of Lake Okeechobee, Florida, where there are many irrigation users of the Floridan Aquifer System. The reservation of the injected waters must be examined in more detail in order to understand the legal issues.

A few of the United States (California and Texas) do not administer water rights for groundwater, and groundwater use is governed by the “correlative rights” legal doctrine which states that landowners may extract as much groundwater as they can put to beneficial use (AWWA, 2002). Texas, California, Florida and Arizona have specific

regulations encouraging owners of ASR to reuse wastewater effluent as part of their projects. Arizona allows users of such water to use the water rights to “extinction” (AWWA, 2002). Several states are considering ASR-specific legislation. The state of Washington has published specific rule changes in Chapter 173-157 of the Washington Administrative Code (McChesney, 2001).

For prospective ASR projects, ownership of the water should be assured through the appropriation of water rights or reservations. Also, site selection set backs should be employed to minimize extraction of stored water by other users. The set back distances and recommendations have been discussed in previous sections of this report.

The social effects of a potential ASR project should be weighed carefully for all projects. Social impacts of water resources projects are commonly investigated by several Federal agencies (U.S. Water Resources Council, 1983). Involving key stakeholders and affected populations in the planning efforts is highly recommended (Delorme et al. 2003). In recent years the water resources planning process has been changing from the construction paradigm to the coordinated or integrated planning approach (Grigg, 1996). ASR planning efforts should take advantage of these societal changes. The current trends point to a more open process involving all key stakeholders. The new approach emphasizes collaborative approaches that include multi-objective planning and issue analysis. Call and Hayes (2001) recommend directly involving the stakeholders in the technical process. The development of social screening criteria can include team building exercises engaging the main stakeholders and public at large. Engaging the stakeholders at an early stage often allows effective “brainstorming” that can assist with the development of the system constraints to be used as screening criteria.

Another recent advance in planning also seems ideally suited to ASR planning efforts. The use of decision support models to involve key stakeholders is becoming widely accepted and utilized. According to CH2M Hill, a decision support model provides many benefits including (CH₂M Hill et al. 2000):

- Helps people get their hands around complex issues
- Allows an apples-to-apples comparison of multiple alternatives
- Reflects key stakeholder concerns and issues
- Provides an objective, structured framework for evaluation
- Utilizes both technical input and policy input
- Defines the trade-offs between competing objectives
- Gets the relevant issues “out in the open”
- Ensures comprehensive evaluation of all relevant performance criteria

CH₂M Hill utilized this methodology successfully in preparing the Monroe County Florida Sanitary Wastewater Master Plan (CH₂M Hill et al. 2000). Their site screening decision model included a number of social considerations. The Monroe County site selection model appears to have much to offer the ASR community as it includes social, environmental and engineering objectives and constraints. For instance, the goal to “maximize public acceptance” could be the same for an ASR well site screening evaluation.

Engineering

The engineering aspects of all ASR projects are complex and diverse. Typically, design engineers or geologists must design the ASR well, recharge/recovery pump, piping, and some type of water treatment system. In addition, operational scenarios must be developed for the project and economic evaluations must be completed in order to determine project feasibility or to optimize an existing project.

For potable ASR projects, the water treatment plant may already exist; for irrigation or in-stream projects, the water treatment system may have to be developed from scratch.

The level of water treatment is usually selected based upon the regulatory requirements along with the project objectives. As was discussed previously, the regulatory water quality requirements for in-stream ASR projects may be more onerous than comparable potable water projects. According to ASCE (2001), the need for treatment should be based upon:

- Physical, chemical, and organic load of the source water
- Ambient ground water quality
- Established and expected ground water quality standards
- Operational considerations
- Specific permit considerations

Treatment systems should be designed to:

- Remove natural organics and suspended solids
- Remove toxics (if any exist)
- Inactivate pathogens through disinfection
- Control pH to minimize scale formation or corrosion potential

Typical systems usually include coagulation/clarification to remove organics and solids, followed by filtration to remove additional solids, followed by disinfection, and then possibly pH adjustment prior to ASR recharge activities. Failure to adequately address the removal of organics and solids can lead to both physical and biological well clogging discussed in Chapter 1 of this report and in earlier sections of this chapter.

Failure to remove toxics may lead to concentrations in excess of one or more regulated parameters. Failure to inactivate pathogens may also lead to a regulatory issue. The use of pH control should be evaluated on a site-by-site basis given the important site-specific issues that need to be studied. In order for the designer to plan the required level of water treatment, guidance values have been established. This report has provided recommended guidance values for TSS, TOC, and various toxics such as heavy metals,

organics, and microcystin. In addition, the report has discussed technical issues associated with disinfection by-products generated from chemical disinfection activities.

The different options to evaluate for disinfection should be considered carefully since each option may create other unforeseen ASR design issues. As described earlier in great detail, chemical chlorination creates disinfection by-products including such compounds as chloroform. The suggested chloroform limit for in-stream ASR projects is very low requiring additional water treatment in order to meet the suggested standard. Chloramination has been shown to produce lower concentrations of disinfection by-products, however, it may lead to production of free ammonia upon ASR recovery. Ammonia can be highly toxic to flora and fauna such as fish or benthic organisms. Ozonation only produces minimal disinfection by-products, however, it produces recharge water that is highly oxygenated. At prospective ASR sites where geochemical issues are a concern, highly oxygenated water can exacerbate the release of heavy metals from some compounds such as pyrite. UV disinfection has not been shown to produce any disinfection by-products, however, since it depends upon light penetration for pathogen inactivation, prior water treatment may be required to remove natural color or TSS. Also, highly alkaline water can lead to fouling of the UV light sources.

ASR operations planning is important to the overall success of the project since it may effect the system performance and the economics. Depending upon the purpose of the project, the timing and duration of operations will determine the operation and maintenance (O&M) cost, the availability and quality of surface water for recharging, automation/instrumentation requirements, and the derived system benefits. For example, in the planning of ASR projects in support of irrigation, the planner must consider the

seasonality of local agricultural users since crop type may dictate when recovered water is needed. Similarly, for in-stream projects, ecosystems have pronounced seasonality in many cases requiring water during specific periods. For potable ASR projects, recharge during off-peak hours could minimize O&M costs as well as possibly minimize impacts upon existing well users who would traditionally utilize their water wells during peak times.

In the case of remote sites located in rural areas, security and nature related problems are likely. For the Blaine Gypsum AR project located in Oklahoma, problems were constant due to the remote rural nature of the project (Bureau of Reclamation, 1998). Problems noted included:

- Fire and lightning damage to equipment
- Flooding of the area
- Damage by animals
- Transducer equipment failures

These considerations should also be factored into the overall cost and design of the ASR facility. Automation of water treatment facilities at remote sites should be secure and reliable using low and high level alarms and automatic shutdown features. In urban areas, vandalism is a key engineering consideration. Vandalism may necessitate installation of security fencing at the ASR project.

Besides general operational decisions that need to be made by the ASR practitioner, water treatment and well maintenance cycles should be considered. One important item that should be planned for in advance is the frequency of well backflushing. In the ASR case histories studied in this report, well backflushing or re-development frequency was mentioned repeatedly due to its perceived importance for maintaining well specific capacities. In a general sense, ASR wells screened within unconsolidated aquifers will

require more attention to periodic backflushing. A regular program maintained at a bi-weekly or daily frequency is recommended. Similar frequencies are recommended for ASR wells located in sandstone or moderately fractured hard rock aquifers. For carbonate sites, weekly to monthly backflushing is probably sufficient depending upon the recharge rate. Sites with higher rates should probably assume a weekly frequency. These recommendations are consistent with those discussed by Pyne (1995).

In the case of brackish water ASR projects, the operations planning is even more important. Poor planning will undoubtedly lead to poor ASR RE percentage, upconing problems, or possibly total failure. A review of the ASR case studies indicates that two general operational strategies have been employed to maximize RE. First, the traditional approach is to develop the ASR storage zone through multiple recharge and recovery cycles. Optimally, a similar size recharge volume is used every cycle to continually buffer the ambient water quality with freshwater recharge. This is a “pore volume” approach consistent with experiments performed by Haggerty et al. (1998). The second approach is establishing a “target storage volume” (TSV) by recharging massive quantities of freshwater for the first few cycles to ensure that a target recovery volume can be achieved as soon as possible. The TSV approach is discussed in more detail by Pyne (1995). For this report, ASR Simulator Model C was utilized to test both strategies assuming a year long simulation with recharge chloride concentration of 50 mg/l and an ambient groundwater chloride concentration of 1,000 mg/l. The total recharge duration was 180 days for both ASR operational strategies and the recharge/recovery rate was maintained at 5 MGD for each.

For the pore volume approach of multiple recharge and recovery cycles, six separate 30 day recharge cycles were simulated using 150 million gallons of recharge for each cycle. ASR recovery for each cycle was started immediately following each recharge cycle, assuming no storage time. The ASR recovery cycle was ended once the chloride concentration at the ASR well exceeded 250 mg/l. In addition, two sets of simulations were developed, one assuming a dispersivity of 1 foot and the other assuming a dispersivity of 25 feet. For strategy one using the dispersivity of 1 foot, 900 million gallons of freshwater were cumulatively recharged and approximately 835 million gallons of freshwater were cumulatively recovered. The recovery volume of the initial cycle was approximately 117,500,000 gallons. The final day of cycle 6 recovery was recorded at day 347 (out of 365). For strategy one using the dispersivity of 25 feet, 900 million gallons of freshwater were cumulatively recharged and approximately 575 million gallons of freshwater were cumulatively recovered. The recovery volume of the initial cycle was approximately 46,250,000 gallons. The final day of cycle 6 recovery was recorded at day 295 (out of 365). Figure 3-5 presents the results for operational strategy # 1.

The model simulations studying the TSV approach utilized the same recharge duration of 180 days with the same recharge rate of 5 MGD. Obviously, only one huge cycle of recharge and recovery was completed. For strategy two using the dispersivity of 1 foot, 900 million gallons of freshwater were cumulatively recharged and approximately 810 million gallons of freshwater were cumulatively recovered. The recovery volume of the initial cycle was approximately 810,000,000 gallons. This recovery volume is 25,000,000 gallons less than the corresponding pore volume approach. The final day of

cycle 1 recovery was recorded at day 342 (out of 365). For strategy two using the dispersivity of 25 feet, 900 million gallons of freshwater were cumulatively recharged

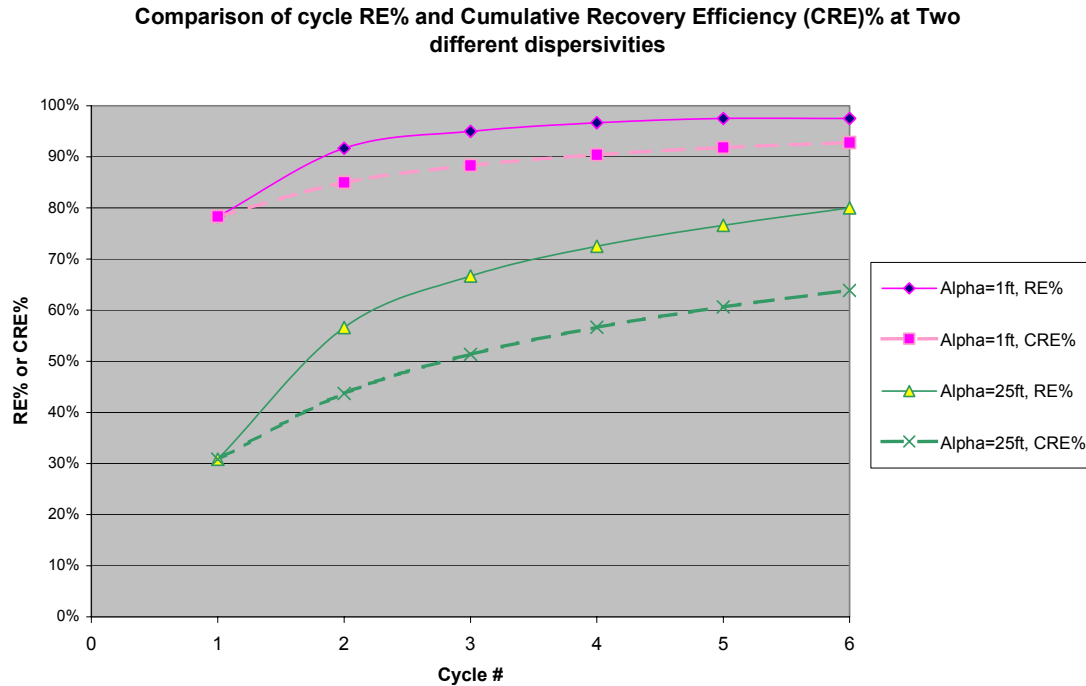


Figure 3-5. ASR performance for operational strategy # 1. RE is recovery efficiency, CRE is the cumulative recovery efficiency, and alpha is the dispersivity.

and approximately 466 million gallons of freshwater were cumulatively recovered. The recovery volume of the initial cycle was approximately 466,200,000 gallons. This recovery volume is 109,000,000 gallons less than the corresponding pore volume approach. The final day of cycle 6 recovery was recorded at day 273 (out of 365).

Each approach has distinct advantages and disadvantages. First, the pore volume approach provides higher cumulative recovery efficiency than the comparable TSV approach. As the aquifer dispersivity increases, this advantage grows considerably. The main disadvantage of the pore volume approach is that the initial recovery volume is much smaller than the comparable TSV initial recovery volume. If recovery water needs dictate a large volume quickly, the TSV approach may be the preferred option. Either

operational approach would require a similar total number of recharge and recovery days with the TSV approach requiring slightly less. Another potential drawback to the TSV approach is the potential scale dependency of the effective dispersivity. As the scale of the ASR storage bubble is increased, the dispersivity would likely rise, further reducing the RE percentage. In the pore volume approach, the effective dispersivity value would be expected to be lower. Further discussion of scale dependent dispersivities is discussed in following sections of this report.

The feasibility of a potential ASR project is defined by many technical factors including project economics. ASR is another water resources option for water storage; in essence ASR functions as an underground storage reservoir. Similarly, ASR is a type of water supply also so it should be compared to other water supply options including wastewater reuse, reverse osmosis or desalinization. During the evaluation of alternatives, unit costs and benefits should be contrasted in order to select the best option for the particular project site (Brown et al. 2005a). According to Brown et al. and the case study summaries developed for this report, the average unit cost of non-brackish water ASR sites is \$0.42 per thousand gallons recovered, while for brackish water ASR sites the mean unit cost is \$0.94 per thousand gallons recovered. These costs compare to unit costs of \$3.00 to \$5.00 per thousand gallons recovered for desalinization, \$1.50 to \$2.00 per thousand gallons recovered for wastewater reuse, and \$0.25 to \$8.00 for surface reservoirs. In many cases, ASR will be the best solution, in some cases another water resources option may be superior. Project costs can be sub-divided into initial capital costs and O&M costs. Capital costs typically include:

- Water treatment system
- Well installation

- Well pump
- Yard pipe
- Surface water intake structure or pipe
- Surface water discharge structure or clear well
- Control buildings
- Appurtenant features including power poles, lighting, security fence

According to Abe (1986), O&M costs can include:

- Project staff and overhead
- Renewals and replacements
- Road maintenance
- Conveyance System O&M
- Recharge facility O&M
- Recovery system O&M

Additional O&M costs can include water treatment costs, security and surveillance costs, disposal of water treatment residuals, and well periodic re-development costs.

The actual economic analysis of the potential ASR system should be completed consistent with recommendations of the U.S. Water Resources Council (1983). For water resources investments undertaken by the United States government, the economic analysis usually involves determination of average annual costs and average annual benefits. For ASR projects under development, the average annual costs include the total average annual O&M costs plus the annualized capital costs. The interest rate to be used for the analysis is published quarterly by the United States government. The investment period should not exceed 100 years, however, 50 years is commonly used (U.S. Water Resources Council, 1983). The value of the recovered water benefit can vary depending upon the intended use of the project. The recovered water may be valued based upon the average price of water per thousand gallons recorded in the study area or it could be linked to the value of the agricultural commodity that is the beneficiary of the water. The value of water for in-stream projects may be considerable depending on the ecological

commodities that are the beneficiaries. An alternative analysis utilizes the total present worth costs and the total present worth value of the recovered water (Bureau of Reclamation, 1994). Ultimately, all water resources projects provide a combination of benefits and costs (Brown et al. 2005a). It would be useful to tie all of the benefits and costs together into an integrated decision framework as is proposed in this research project. Also, once the decision framework has been developed, it should be automated in the future.

The use of automated decision support models combining economic, engineering, and scientific criteria is becoming widely accepted and utilized. It appears that the public sector has embraced the use of computer-based decision models to assist with every thing from highway planning to urban sprawl (Saunders-Newton, 2001; Johnson, 2001). Private companies also are taking advantage of decision support models to estimate optimal commodity delivery. The natural gas industry has been using these types of models in one form or another for over 10 years (Chin and Vollman, 1992). The use of decision support models in support of ASR planning may well be based upon the work completed for this research effort.

Special Considerations Specific for Brackish Water ASR Sites

In addition to guidelines and metrics developed for all ASR projects, brackish water ASR projects are subject to additional planning and design considerations. Both guidance values and performance metrics have been developed as part of this research report. The planning guidance values are based upon previous research completed by others and new modeling completed in this report and described in Chapter 2. Multiple ASR planning factors and variables have been evaluated for this research effort. Each performance variable is presented in turn and a planning guidance value is proposed for

each. The ASR brackish water performance metrics were developed specifically for this report and are discussed in subsequent sections of this report.

Thickness of storage zone

Thickness of the storage zone does affect the recovery efficiency performance.

This has been determined through previous research and through the extensive modeling performed for this research effort. Table 3-11 provides a summary of the study results from previous research as well as this report.

Table 3-11. Research results relating ASR recovery efficiency versus aquifer storage zone thickness (b)

Variable	Thin storage zones provide higher RE	Thin storage zones have no effect on RE	Reference or Source
Storage zone b	X	----	Kimbler et al. (1975)
Storage zone b	X	----	Khanal (1980)
Storage zone b	X	----	Merritt (1986)
Storage zone b	X	----	Streetly (1998)
Storage zone b	X	----	This Report

The recommended minimum thickness (25 feet) of an unconfined aquifer has been discussed in previous sections of this report. For confined brackish water storage zones, an optimum storage zone thickness should be between 50 and 100 feet. At storage zone thickness of less than 50 feet, developed recharge pressures could be excessive. Storage zone thickness between 50 and 100 feet should result in satisfactory recovery efficiency and provide operational flexibility to consider high recharge rates since larger aquifer thickness is directly proportional to higher aquifer transmissivity.

Effect of aquifer regional pre-existing gradient

Pre-existing aquifer groundwater gradient does affect the recovery efficiency performance. This has been determined through previous research and through the

extensive modeling performed for this research effort. Table 3-12 provides a summary of the study results from previous research as well as this report.

Table 3-12. Research results relating ASR recovery efficiency versus pre-existing aquifer gradient (i)

Variable	Lower gradients provide higher RE	Gradients have no effect on RE	Reference or Source
Gradient i	X	----	Harpaz and Bear (1964)
Gradient i	X	----	Kimbler et al. (1975)
Gradient i	X	----	Merritt (1986)
Gradient i	X	----	Anderson and Lowry (2004)
Gradient i	X	----	This Report

Based upon the modeling performed in this report, an ideal ASR site would have a gradient less than 0.001. Prospective ASR sites with gradients larger than this may not be good candidate ASR sites, especially if the ambient groundwater quality is poor. This guidance value is discussed under the “hydraulics” section of this report but is re-emphasized here since the gradient is more critical to RE performance in brackish water.

Recharge volume

Recharge volume does affect the recovery efficiency performance. This has been determined through previous research and through the extensive modeling performed for this research effort. Table 3-13 provides a summary of the study results from previous research as well as this report.

Table 3-13. Research results relating ASR recovery efficiency versus recharge volume

Variable	Higher recharge volumes provide higher RE	Recharge volume has no effect on RE	Reference or Source
Recharge Volume	X	----	Merritt (1986)
Recharge Volume	X	----	Streetly (1998)
Recharge Volume	X	----	Pavelic et al. (2002)
Recharge Volume	X	----	Anderson and Lowry (2004)
Recharge Volume	X	----	This Report

Based upon the previous research and modeling performed in this report, the ideal initial recharge volume to maximize RE ranges from 40 to 70 million gallons depending upon the amount of hydrodynamic dispersion. Aquifer storage zones characterized by low dispersion only need to recharge 40 million gallons to maximize RE, whereas, aquifers characterized by high dispersion may invest over 70 million gallons to maximize recovery efficiency. The effect of hydrodynamic dispersion is discussed below.

Dispersivity

The degree of hydrodynamic dispersion as measured through the intrinsic dispersivity does affect the recovery efficiency performance. This has been determined through previous research and through the extensive modeling performed for this research effort. Table 3-14 provides a summary of the study results.

Table 3-14. Research results relating ASR recovery efficiency versus dispersivity

Variable	Low dispersivity provide higher RE	Dispersivity has no effect on RE	Reference or Source
Dispersivity	X	----	Harpaz and Bear (1964)
Dispersivity	X	----	Esmail (1966)
Dispersivity	X	----	Kimbler et al. (1975)
Dispersivity	X	----	Reeder et al. (1976)
Dispersivity	X	----	Merritt (1986)
Dispersivity	X	----	Yobbi (1996)
Dispersivity	X	----	Streetly (1998)
Dispersivity	X	----	Pavelic et al. (2002)
Dispersivity	X	----	Missimer et al. (2002)
Dispersivity	X	----	Anderson and Lowry (2004)
Dispersivity	X	----	This Report

Based upon the previous research and modeling performed in this report, dispersivity values should be minimized to ensure economical RE for brackish water ASR projects. Although further research is warranted, the geological character of the aquifer storage

zone appears to be related to the dispersivity. This conclusion is consistent with recent research by Schulze-Makuch (2005) that determined a power law relationship relating geologic character and project scale to dispersivity. Equation 6 provides the general relationship determined by the research.

$$\text{Equation (6)} \quad \alpha = cL^m$$

where c is a characteristic parameter of a geologic medium; L is the flow distance or problem scale; and m is a scaling exponent related to the geologic medium.

Harpaz (1971) showed that sandstones are likely to yield higher recovery efficiencies than limestones. Reeder et al. (1976) noted that tracer tests in dolomite provided an estimate of dispersivity of 280 feet. Brown and Silvey (1977) discuss injection tests into a sandy aquifer in Virginia. In general, RE was high but hampered by geochemical problems. Merritt et al. (1983) discuss an ASR test near Tampa, Florida in a limestone storage zone with karst zones. They speculate that high hydrodynamic dispersion in the karst zones led to poor ASR RE. At the Cocoa Beach ASR site, Pyne (1995) mentions that well # R2 is affected by karst-like conduits that effectively reduce the RE percentage due to mixing and dispersion. Streetly (1998) found that using a dispersivity of 131 feet significantly reduced the RE in model simulations. Missimer et al. (2002) noted that high permeability zones in a limestone storage zone could lead to poor recovery efficiency due to increased mixing. Sibenaler et al. (2002) speculate that karst conduit zones at the Willunga ASR site may have resulted in poor RE.

If ASR storage zone geologies are considered as a continuum from mostly porous media character to dual-porosity aquifer character, it is possible to order them from lowest to highest dispersivity. From the literature reviewed in this report and from an

evaluation of the case studies developed and the modeling completed, the amount of hydrodynamic dispersion can be generally ordered as follows in Tables 3-15 and 3-16.

The tables are sorted as unconsolidated sediments and consolidated rocks. Further support for this hypothesis is discussed in subsequent sections of this report.

Table 3-15. General unconsolidated aquifer character versus degree of dispersion expected during ASR recharge and recovery

Aquifer character	Fine to medium Sand	Medium to coarse Sand	Medium to coarse Sand – highly heterogeneous	Fine to medium Gravel	Medium to coarse Gravel	Medium to coarse Gravel – highly heterogeneous
Degree of dispersion	Low	Low to Moderate	Moderate	Moderate	Moderate to High	High

Table 3-16. General rock aquifer character versus degree of dispersion expected during ASR recharge and recovery

Aquifer character	Sand stone	Mod. fractured Sandstone	Sandy Lime stone	Limestone with minimal conduits	Frac. rock	Karst Lime stone	High fractured rock	Dolomite or Chalk or Karst Limestone with major caverns
Degree of dispersion	Low	Low to Mod	Low	Moderate to High	Mod	High	High	Very High

In order to assign dispersivity ranges to each one of the aquifer character types, a review of published dispersivity values along with ASR dispersivity information was completed. Much of the data compiled was contained in literature or provided by fellow ASR researchers. Pavelic et al. (2002) developed estimates of dispersivity of various ASR sites in Australia and around the world using analytical solutions. Discussions were held with a number of local experts from the State of Florida. Dr. Mark Pearce of the Florida consulting firm Water Resources Solutions provided additional information.² Many other studies were reviewed for their applicability to this research report. Although

² Personal communication on 4/28/2005

the data was not collected at ASR project sites, additional dispersivity estimates have been extracted from literature reviewed (Reeder et al. 1976; Baker, 1977; Chiang et al. 1989; Steele et al. 1989; Tsang et al. 1991; Maloszewski et al. 1992; Moltyaner et al. 1993; Ptak and Teutsch, 1994; Schulze-Makuch, 1996; Merritt, 1997; D'Alessandro et al. 1997; Domenico and Schwartz, 1998) and ASR case studies developed for this report.

Table 3-17 compares the estimated dispersivity values versus the predominant geologic character.

Table 3-17. General aquifer character versus estimated dispersivity

ASR Project Site or Literature Study Site	Predominant Geologic Character	Estimated Dispersivity (feet)
Unconsolidated site reported on by Moltyaner et al. 1993	Sandy sediments	0.53 to 1.54
Las Vegas, USA	Cemented sands and gravels	13.12
East Bay, USA	Alluvial sands and gravels	16.41
Calleguas, USA	Marine sands and gravels	22.0
Unconsolidated site reported on by Ptak and Teutsch, 1994	Alluvial sand and gravel deposits	1.69 to 30.91
Memphis, USA	Coarse sand and gravel	98.43
Unconsolidated site reported on by Chiang et al. 1989	Coarse sand, gravel, and cobbles	28.10
Wisconsin site reported on by Schulze-Makuch, 1996	Sandstone	0.0042 to 0.005
Charleston, USA	Limestone/sandstone	0.33
Waruwi, Aus	Sandstone	1.97
Jandakot, Aus	Sandstone	65.62
Mawson Lakes, Aus	Sandy limestone	2.62
Bolivar, Aus	Sandy limestone	6.5 to 16.5
Tampa Rome Avenue, USA	Sandy limestone	20.0
Core sample tests reported on by Baker, 1977	Vugular limestone	3.96 to 7.84

Table 3-17. Continued.

ASR Project Site or Literature Study Site	Predominant Geologic Character	Estimated Dispersivity (feet)
Marco Lakes, USA	Limestone	30.0
Lee County Olga, USA	Limestone	39.0
Hialeah, USA	Limestone	65.0
Lee County North, USA	Limestone	78.0 considered poor estimated by M. Pearce
Site reported on by Maloszewski et al. 1992	Karst Limestone	33.4 to 136.8
Andrews Farm, Aus	Sandy limestone with conduits	82.03
Willunga, Aus	Sandy limestone with conduits	164.0
Washington site reported on by Steele et al. 1989	Fractured basalt	0.56 to 14.0
Site reported on by D'Alessandro et al. 1997	Fractured granite	9.55
Site reported on by Tsang et al. 1991	Fractured granite	1.7 to 19.1
Minnesota AR site, USA	Dolomite	280.0
USA fractured rock experiment site from Domenico and Schwartz, 1998.	Fractured rock	440.0

Based upon the compiled data, the range of estimated dispersivity versus predominant geology for all sites is presented in Tables 3-18 and 3-19. Further support for this hypothesis is discussed in subsequent sections of this report. For ASR planning purposes, guidance values based on this data are also presented on Tables 3-18 and 3-19. The guidance values are thought to be reasonable values for feasibility-level planning purposes. Obviously, if in-situ dispersivity can be determined through the use of tracer tests or other field exploration technique, those values should be used instead. The suggested guidance values shown in both tables tend to be within the lower third of the range listed for most values. This was assumed since if geologic character is related to dispersivity, it may also be a loose function of the hydraulic conductivity

Table 3-18. General unconsolidated aquifer character versus dispersivity range expected during ASR recharge and recovery

Aquifer character	Fine to medium Sand	Medium to coarse Sand	Medium to coarse Sand – highly heterogeneous	Fine to medium Gravel	Medium to coarse Gravel	Medium to coarse Gravel – highly heterogeneous
Degree of dispersion	Low	Low to Moderate	Moderate	Moderate	Moderate to High	High
Dispersivity Range	0.53 to 1.54	1.69 to 22	22 to 30.91	22 to 30.91	30.91 to 98.43	98.43
Guidance Value for planning efforts (in feet)	0.50	5.0	25.0	25.0	50.0	75.0

Table 3-19. General rock aquifer character versus dispersivity range expected during ASR recharge and recovery

Aquifer character	Sand stone	Mod. fractured Sandstone	Sandy Lime stone	Limestone with minimal conduits	Frac. rock	Karst Lime stone	High fractured rock	Dolomite or Chalk or Karst Limestone with major caverns
Degree of dispersion	Low	Low to Mod	Low	Moderate to High	Mod	High	High	Very High
Dispersivity Range	.004 to 1.97	1.97 to 65.62	2.62 to 20	3.96 to 78.0	0.56 to 9.55	33.4 to 164	9.55 to 19.1	280 to 440
Guidance Value for planning efforts (in feet)	0.50	10.0	7.5	30.0	3.0	65.0	12.5	100.0

distribution in the aquifer. Typically, the best stochastic model distribution to represent conductivity is log-normal. The maximum value recommended is 100 feet since Gelhar et al. (1992) suggest that studies yielding high dispersivities (e.g. 280 and 440 feet) are generally less reliable than other field-derived values. In addition to the table values suggested above, it may be useful to use Equation 6 [Schulze-Makuch (2005)] presented above to develop a planning estimate of dispersivity. The analytical approach proposed

by Schulze-Makuch also incorporates project scale into the dispersivity estimate; this idea is more inline with the conventional wisdom on the subject. Obviously, a scale factor in terms of ASR projects can be thought of as the theoretical storage volume radius. To use this approach, it is assumed that the dispersivity can be correlated to scale, but is not totally dependent upon it. Both approaches were utilized in model validation.

ASR storage duration

The storage duration of the recharged water does affect the recovery efficiency performance during recovery operations. Generally, long storage periods reduce the overall RE percentage. This reduction is due to a combination of pre-existing gradient translating the stored water away from the well and buoyancy stratification due to density differentials between recharge water and ambient groundwater. In addition, the compilation of case study information completed for this report revealed real field examples of this phenomenon. Table 3-20 provides a summary of the study results from previous research as well as this report.

Table 3-20. Research results relating ASR recovery efficiency versus storage duration

Variable	Limited storage durations of less than 90 days provide higher RE	Storage duration has no or limited effect on RE	Reference or Source
Storage duration	X	----	Harpaz and Bear (1964)
Storage duration	X	----	Khanal (1980)
Storage duration	X	----	Merritt (1986)
Storage duration	----	X **	Streetly (1998)
Storage duration	X	----	Pavelic et al. (2002)
Storage duration	X	----	Missimer et al. (2002)
Storage duration	X	----	Anderson and Lowry (2004)
Storage duration	X	----	This Report

Note - ** Streetly only evaluated up to 30 days of storage

Based upon the previous research and new research developed for this report, storage durations should be kept to a minimum. At modest density differentials where the recharge water is freshwater and the ambient groundwater contains up to 2,500 mg/l chloride or up to 5,000 mg/l TDS, storage durations of less than 180 days should be adequate to ensure satisfactory RE percentage. In cases where the ambient groundwater has a TDS value between 5,000 and 10,000 mg/l, storage durations should be kept to less than 90 days. Obviously, projects that can operate with no storage period would ensure maximum performance of the ASR wells.

Density of ambient groundwater in storage zone

The density of the ambient groundwater as compared to the recharge water does affect the recovery efficiency performance during recovery operations. Generally, large density differentials lead to pronounced buoyancy stratification in the ASR storage zone. This has been determined through previous research and through the extensive modeling performed for this research effort. In addition, the compilation of case study information completed for this report revealed real field examples of this phenomenon. Table 3-21 provides a summary of the study results from previous research as well as this report.

Table 3-21. Research results relating ASR recovery efficiency versus density differential

Variable	Limiting TDS in ASR storage zone to 10,000 or less provides higher RE	TDS has no or limited effect on RE	Reference or Source
Density differential	X	----	Kimbler et al. (1975)
Density differential	X	----	Khanal (1980)
Density differential	X	----	Merritt (1986)
Density differential	X	----	Yobbi (1996)
Density differential	X	----	Missimer et al. (2002)
Density differential	X	----	This Report

Based upon the previous research and new research developed for this report, the density differential should be minimized in order to ensure economical RE percentage. Generally, the ambient groundwater should have a TDS value of less than 10,000 mg/l to ensure project feasibility. Two notable projects (Marathon, Florida [Pyne, 1995] and Clayton, Australia [Gerges et al. 2002b]) have demonstrated that storing water in aquifers containing sea water is possible but only for limited emergency use. For ASR projects intended for extended water supply use or for large in-stream projects, limiting the ambient groundwater TDS to less than 10,000 mg/l is recommended.

Recharge and ambient groundwater water quality

The ambient groundwater quality as compared to the recharge water quality does affect the recovery efficiency performance during recovery operations due to mixing of the two different waters. Besides the issue of density differential, high TDS and chloride levels in the ambient groundwater will lead to poor RE due to simple mixing. Higher TDS or chloride levels in the recharge water will also lead to poor RE since the recharge water quality may already be near the regulatory limit for TDS or chloride. The recovered water will be a mixture of the recharge and ambient water; therefore, the poorer the initial water quality for each, the lower the RE percentage to be expected. This has been determined through previous research and through the extensive modeling performed for this research effort. In addition, the compilation of case study information completed for this report has revealed numerous real field examples of this phenomenon. In fact most brackish water ASR projects exhibited this relationship. Table 3-22 provides a summary of the study results from previous research as well as this report. The results clearly show that all previous authors agree on the effect of poor ambient water quality.

Table 3-22. Research results relating ASR recovery efficiency versus ambient groundwater quality

Variable	Lower RWQ or AWQ provide higher RE	RWQ or AWQ has no or limited effect on RE	Reference or Source
Recharge or ambient water quality	X	----	Kimbler et al. (1975)
Recharge or ambient water quality	X	----	Khanal (1980)
Recharge or ambient water quality	X	----	Merritt (1986)
Recharge or ambient water quality	X	----	Streetly (1998)
Recharge or ambient water quality	X	----	This Report

RWQ is recharge water quality; AWQ is ambient groundwater quality

Due to storage duration and density differential, the TDS of the ambient groundwater has been recommended to be less than 10,000 mg/l. To ensure that the recovered water quality is reasonable and meets the required TDS or chloride limits discussed previously in this report, it is recommended that the TDS of the ambient groundwater be limited to less than 5,000 mg/l while the chloride be limited to less than 2,500 mg/l. For recharge water, it is recommended that the TDS of the ambient groundwater be limited to less than 400 mg/l while the chloride be limited to less than 200 mg/l. Using these guideline values, most ASR projects can be designed to be technically viable and economically feasible. Obviously, some ASR projects may be feasible at water quality levels worse than these guideline values, however, the economics of such projects will be less than optimum.

Effect of multiple consecutive cycles

The RE of brackish water ASR sites has been shown to improve with increasing number of recharge and recovery cycles. This has been determined through previous research and through the extensive modeling performed for this research effort. In addition, the compilation of case study information completed for this report revealed real field examples of this phenomenon. Table 3-23 provides a summary of the study results from previous research as well as this report.

Table 3-23. Research results relating ASR recovery efficiency versus number of recharge and recovery cycles

Variable	RE improves with increasing number of cycles	The number of cycles has no or limited effect on RE	Reference or Source
Number of recharge and recovery cycles	X	----	Kimbler et al. (1975)
Number of recharge and recovery cycles	X	----	Merritt (1986)
Number of recharge and recovery cycles	X	----	Pyne (1995)
Number of recharge and recovery cycles	X	----	Yobbi (1996)
Number of recharge and recovery cycles	X	----	Streetly (1998)
Number of recharge and recovery cycles	X	----	This Report

The RE percentage does increase as the number of cycles increase, however, the rate of increase is a function of the ambient groundwater quality and the degree of hydrodynamic dispersion. In addition, optimum performance is assured through the use of moderate recharge and recovery volumes. A recharge volume of 20 million gallons at a minimum is recommended for operational purposes. At sites with TDS less than 2,000 mg/l or chloride concentrations less than 1,000 mg/l and moderate dispersivity of 50 feet, four to six recharge and recovery cycles of at least 20 million gallons should result in RE

percentages greater than 70%. At sites with TDS less than 6,000 mg/l or chloride concentrations less than 3,000 mg/l and moderate dispersivity of 50 feet, eight to ten recharge and recovery cycles should result in RE percentages greater than 70%.

Obviously, some ASR projects may be feasible at water quality levels worse than these guideline values, however, the economics of such projects will be less than optimum, since a minimum of ten recharge and recovery cycles is recommended to improve RE percentages to greater than 70%.

Aquifer homogeneity/isotropy

The RE of brackish water ASR sites has been shown to vary with the degree of homogeneity and the degree of anisotropy. Unfortunately, previous research and modeling completed for this research effort do not entirely agree.

Previous research does not reveal the true effect of either homogeneity or isotropy on brackish water ASR RE percentage. Table 3-24 provides a summary of the study results from previous research as well as this report for the degree of homogeneity.

Additional research on the effect of homogeneity on RE percentage is probably warranted given the limited information available. More research is available concerning the effect of isotropy of an aquifer and its effect on RE percentage. Table 3-25 provides a summary of the study results from previous research as well as this report for the degree of

Table 3-24. Research results relating ASR recovery efficiency versus degree of aquifer homogeneity

Variable	RE worsens with increasing degree of heterogeneity	The degree of heterogeneity has no or limited effect on RE	Reference or Source
Degree of homogeneity	X	----	Huntley and Bottcher (1997)
Degree of homogeneity	X	----	This Report

isotropy. It is evident from the summary table that the effect of anisotropy on RE percentage is not well understood yet and that additional research is warranted.

Table 3-25. Research results relating ASR recovery efficiency versus degree of aquifer isotropy

Variable	RE worsens with increasing degree of anisotropy	The degree of anisotropy has no or limited effect on RE	Reference or Source
Degree of isotropy	X	X	Merritt (1986) *
Degree of isotropy	X	----	Huntley and Bottcher (1997)
Degree of isotropy	----	X	Yobbi (1996)
Degree of isotropy	X	----	Missimer et al. (2002) **
Degree of isotropy	----	X	This Report

Notes - * depends upon the amount of dispersion; ** as anisotropy increases, density stratification actually decreases

Dipping storage zones

The effect of ASR storage zone geologic dip has only been investigated by Kimbler et al. (1975). Kimbler et al. found that aquifer storage zones with geologic dips of 30 degrees or greater reduced RE percentage in a sea water aquifer. The aquifer dip resulted in the stored freshwater moving up dip due to density differentials. Although no additional modeling of this attribute was completed for this report, it is recommended that ASR storage intervals be selected in areas where aquifer dip is at a minimum. In fact, geologic dome-like storage areas may be optimal storage zones.

Aquifer transmissivity

The overall effect of aquifer transmissivity on RE percentage is not entirely known at this time. Research efforts have provided contradictory results. The results of previous research and the extensive modeling performed for this research effort are compiled in Table 3-26.

Table 3-26. Research results relating ASR recovery efficiency versus aquifer transmissivity

Variable	RE decreases with increasing transmissivity	The transmissivity has no or limited effect on RE	Reference or Source
Aquifer transmissivity	X	----	Khanal (1980)
Aquifer transmissivity	X	----	Merritt (1986)
Aquifer transmissivity	X	----	Yobbi (1996) *
Aquifer transmissivity	----	X	Streetly (1998)
Aquifer transmissivity	X	----	Missimer et al. (2002)
Aquifer transmissivity	----	X	This Report

Notes - * Yobbi noted that RE decreased with increasing hydraulic conductivity

The summary table does not provide a definitive conclusion regarding the effect of aquifer transmissivity on RE percentage. Modeling for this research report has shown that at low to moderate density differential of the recharge water and ambient groundwater, aquifer transmissivity has only a minimal effect on RE percentage. Previous research has shown that in cases of high density differential such as storing fresh water in a sea water aquifer, higher transmissivities lead to more intensive buoyancy stratification. Therefore, it is recommended that ASR projects seek storage intervals with moderate transmissivities from 1,000 to 50,000 ft²/day. Aquifers with higher transmissivities are generally associated with karst-like conditions or highly fractured bedrock where dispersivities would be expected to be very high, leading to poor RE.

Aquifer porosity

Aquifer porosity does not affect the recovery efficiency performance very much. This has been determined through previous research and through the extensive modeling

performed for this research effort. Table 3-27 provides a summary of the study results from previous research as well as this report. Research completed for this report indicates that RE is not sensitive to porosity unless values are less than 5%. At low porosities, groundwater velocities are higher leading to faster translation of stored freshwater. Since aquifer effective porosity is sometimes difficult to measure, aquifer geologic character may be a better indicator of ASR performance.

Table 3-27. Research results relating ASR recovery efficiency versus aquifer porosity

Variable	Aquifer Porosity greatly affects RE	Aquifer Porosity only has a minimal effect on RE	Reference or Source
Porosity	----	X	Merritt (1986)
Porosity	----	X	Yobbi (1996)
Porosity	X	----	Anderson and Lowry (2004) *
Porosity	----	X	This Report

Notes - * Anderson and Lowry found low values of porosity combined with high groundwater velocities resulted in reduction of RE

Brackish Water Performance Metrics

Based upon the literature reviewed and the modeling work completed for this research report, it was hypothesized that simple dimensionless parameters could be utilized to predict the initial ASR RE percentage of cycle one. Then, relationships involving the cumulative recovery efficiency would be utilized to predict the long-term performance of the prospective ASR project site. The predictions would be utilized to conduct initial feasibility level investigations of multiple prospective ASR project sites to ensure that exploration and pilot project expenditures are warranted.

The development of the dimensionless performance parameters utilized a varied combination of different performance variables discussed previously in this report. The

literature has revealed and this research report has confirmed, that the most important ASR performance variables are:

- Hydrodynamic dispersion (e.g., dispersivity)
- Recharge Volume and resulting freshwater “bubble size”
- Recharge water quality
- Ambient groundwater quality

The assumptions for allowing valid use of the new dimensionless parameters include the following:

- The ambient groundwater TDS concentration is less than 5,000 mg/l and the chloride concentration is less than 2,500 mg/l
- The ASR storage duration between recharge and recovery is less than 90 days
- The freshwater to be stored has a chloride concentration of less than 250 mg/l
- The aquifer can be approximated as a porous media

Based upon these observations, two different dimensionless parameters were developed and then utilized to formulate theoretical ASR performance envelopes. The first dimensionless parameter relates the recharge water quality to the ambient groundwater quality. The second dimensionless parameter relates the theoretical radius of the freshwater bubble to the hydrodynamic dispersivity. The two dimensionless parameters are discussed in the following sections of the report.

Dimensionless recovery index

As has been noted previously, the Recovery Index is a simple dimensionless parameter relating the recharge water quality and the ambient groundwater quality normalized against the water quality regulatory level. Chloride was selected as the water quality indicator parameter since a majority of the ASR projects identified are utilized for potable water supply where the regulatory driver is usually chloride. Alternate parameters such as TDS were also considered but rejected since TDS can be markedly affected by dissolution in carbonate aquifers. TDS should be considered if ASR

irrigation use is the only project purpose. Other water quality parameters could also be used in the index but were not selected due to the propensity of chloride data available at the sites studied. For this report, the Recovery Index is defined using chloride as the regulatory parameter of choice. Equation 7 provides the numerical definition of the Recovery Index.

$$\text{Equation (7)} \quad \text{AWQ} / (\text{WQ regulatory limit} - \text{RWQ})$$

where the AWQ is the ambient groundwater chloride concentration; WQ regulatory limit is the water quality indicator regulatory limit, in this case chloride with a limit of 250 mg/l; and RWQ is the recharge water chloride concentration. This new dimensionless parameter was then calculated at ASR project sites reviewed for this report. The values calculated in this research report represent average water quality for both AWQ and RWQ. It is recognized that these values may vary temporally during an ASR recharge or recovery event. This is especially true for project sites utilize source water from a lake, river, canal or other natural body of surface water.

Figure 3-6 details the results of the calculation for 15 “high recharge rate” brackish water ASR sites reviewed in this report where dependable data was readily available or collected by the author for this report. The figure also depicts the logarithmic best-fit curve along with the correlation coefficient. The r^2 coefficient is 0.66; since this is greater than 0.50, it suggests an important relationship between Recovery Index (RI) and recovery efficiency (RE%). Figure 3-7 shows similar trends for ASR sites with recharge rates less than 0.5 MGD except that the r^2 coefficient is slightly higher at a value of 0.71.

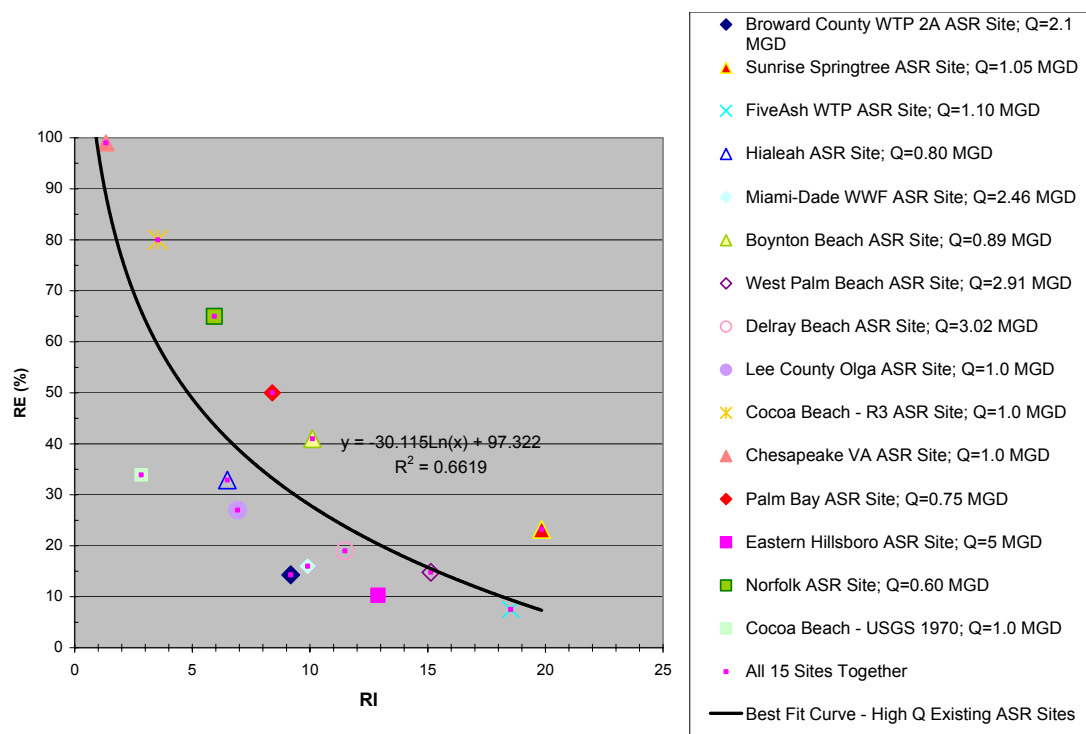


Figure 3-6. Recovery Index (RI) versus Cycle one recovery efficiency (RE%) for brackish water ASR sites where recharge rate was greater than 0.5 MGD and less than 3 MGD

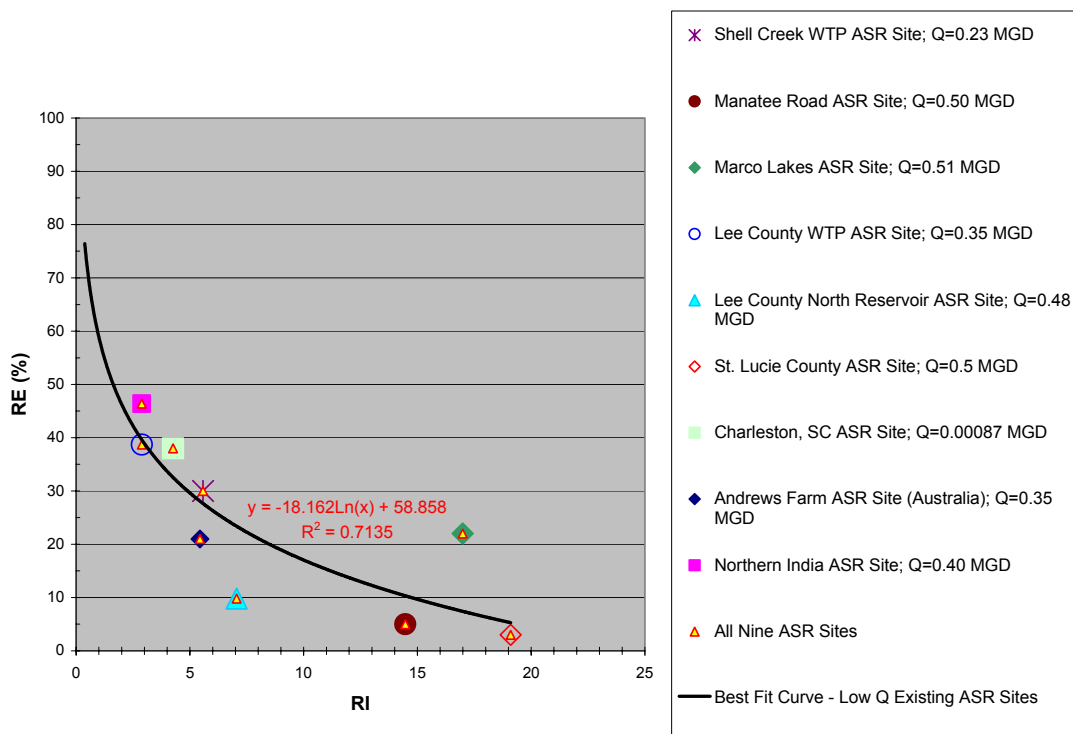


Figure 3-7. Recovery Index (RI) versus Cycle one recovery efficiency (RE%) for brackish water ASR sites where recharge rate was less than 0.5 MGD

The field data suggest that the relationship between RI and RE percentage is a logarithmic one. The field data comparisons of the two figures also reveal that the high volume recharge sites generally have higher overall cycle one RE values suggesting again that recharge volume is another important variable to consider in the analysis. This observation is further reinforced if the Marco Lakes data from Figure 3-7 is thoroughly evaluated. This site appears as a slight data outlier on Figure 3-7, however, when the cycle one recharge volume of this site (e.g., almost 20 million gallons) was studied, as compared to other sites of comparable RI such as St. Lucie County and Manatee Road, it was found that the recharge volume of St. Lucie County was one tenth that of Marco Lakes while the recharge volume of Manatee Road was three times less. A similar finding was discovered on Figure 3-6 when comparing Sunrise Springtree ASR versus Fiveash ASR. Sunrise Springtree had an initial cycle one recharge volume almost twice the size of Fiveash. In addition to the water quality dimensionless parameter developed for this report, another dimensionless parameter incorporating dispersivity is required in order to provide meaningful performance metrics since hydrodynamic dispersion has been shown to be the single most important performance variable for ASR projects. The following section describes the development and use of the inverse relative dispersivity.

Dimensionless relative dispersivity

The relative dispersivity is a concept that was introduced by Pavelic et al. (2002) as a way to relate the size of the stored freshwater bubble to the aquifer intrinsic dispersivity. Equation 8 provides the mathematical definition of the relative dispersivity.

$$\text{Equation (8)} \quad R_{\alpha} = \alpha / r_m$$

where α is the intrinsic dispersivity measured in feet and r_m is the theoretical radius of the stored water bubble represented as an idealized cylinder (also measured in feet).

Obviously, r_m itself is a function of the aquifer effective porosity (η), the aquifer thickness (b), and the recharge volume (V). Equation 9 provides the mathematical definition of r_m .

$$\text{Equation (9)} \quad r_m = [(V / (\pi\eta b))]^{1/2}$$

Typically, dispersivities are much lower numerically than the cylinder radius, so for this report, the inverse of the relative dispersivity has been utilized, and is identified as R_α' . The use of the inverse ensures that values for the inverse relative dispersivity are larger numbers. In addition, typically for ASR projects, larger values of inverse relative dispersivity are better than lower values so logically an ASR practitioner would seek to maximize this dimensionless parameter. The inverse relative dispersivity appears to be an ideal dimensionless parameter for ASR projects based upon the literature and modeling completed for this report since it relates the other two key performance variables (e.g., recharge volume and hydrodynamic dispersion), however, the intrinsic dispersivity is usually not known at the feasibility level of investigations. Therefore, the user would have to rely upon literature data or guideline values provided previously in this report unless better estimates of dispersivity can be developed based upon real ASR field data. In addition to the real field data that could be analyzed, numerical modeling could also provide more theoretical ASR performance data that could be evaluated. The subsequent sections describe the process completed for this report that relates numerical model results with the two proposed dimensionless parameters.

Compilation of model simulations

Using ASR model simulators A and C, discussed in Chapter 2 of this report, a series of over 500 separate model runs were completed in order to evaluate the effect of the main performance variables discussed above. In essence, the numerical model results were utilized to create a “synthetic” data set that could be evaluated further using multiple parameter regression modeling. The model runs varied the following parameters:

- Recharge water quality (RWQ)
- Ambient groundwater quality (AWQ)
- Dispersivity (α)
- Transmissivity (T)
- Recharge Volume (V)
- Recharge Rate (Q)

These variables were selected based upon the initial parameter sensitivity tests completed to support the assessment of ASR performance variables discussed in Chapter 2. The modeling completed for Chapter 2 evaluated a much larger number of parameters but indicated that the above listed variables were the most important. In addition, further development of dimensionless parameters discussed in the previous section required model simulations varying RWQ, AWQ, α , and V. Although it has been recognized that the porosity is also a minor variable of interest, it was maintained as a constant 25% throughout all of the simulations. The transmissivity, T, was also varied in the simulations since previous work and modeling completed in support of Chapter 2 provide contradictory evidence of its overall importance. The recharge rate, Q, was varied in some cases to allow for shorter model simulations (e.g., higher rates were used for a shorter duration in some instances).

Each simulation included 10 to 30 days of recharge with immediate recovery for the same duration. The recovery RE percentage was calculated once the simulated chloride concentration at the ASR well exceeded 250 mg/l. None of the simulations included storage time since it has previously been established that storage durations will reduce the potential recovery efficiencies and that density differentials were more important during these instances as compared to operations with no storage period. Also, ASR simulator B was only utilized for a limited number of model simulations to evaluate potential errors in RE estimates produced using a non-density dependent model. It is important to point out that using ASR simulator B for all model runs was considered but rejected once the computer CPU time required was calculated. Also, as demonstrated in Chapter 2, the RE estimate errors produced using a non-density dependent model are very small when storage durations are kept to a minimum (e.g., 2 to 8%). One other important assumption to note for the simulations is one regarding the dispersivity tensor. Generally, numerical models allow input of longitudinal, transverse, and vertical dispersivities. Previous research has shown that of the three dispersivities, the longitudinal is the largest and the vertical value is the smallest. For this report, it was assumed that all dispersivities were equal. This assumption is probably conservative and possibly biased the simulation results to a small degree. The potential bias is discussed in later sections of this study.

Figure 3-8 displays time versus chloride concentration data for select ASR projects discussed in this research report. The shapes of the recovery curves are a function of the RWQ, AWQ, V, and α . It is obvious to see that the conditions at each project site vary considerably. Figure 3-9 displays some of the same field data along with simulation results from ASR simulator A.

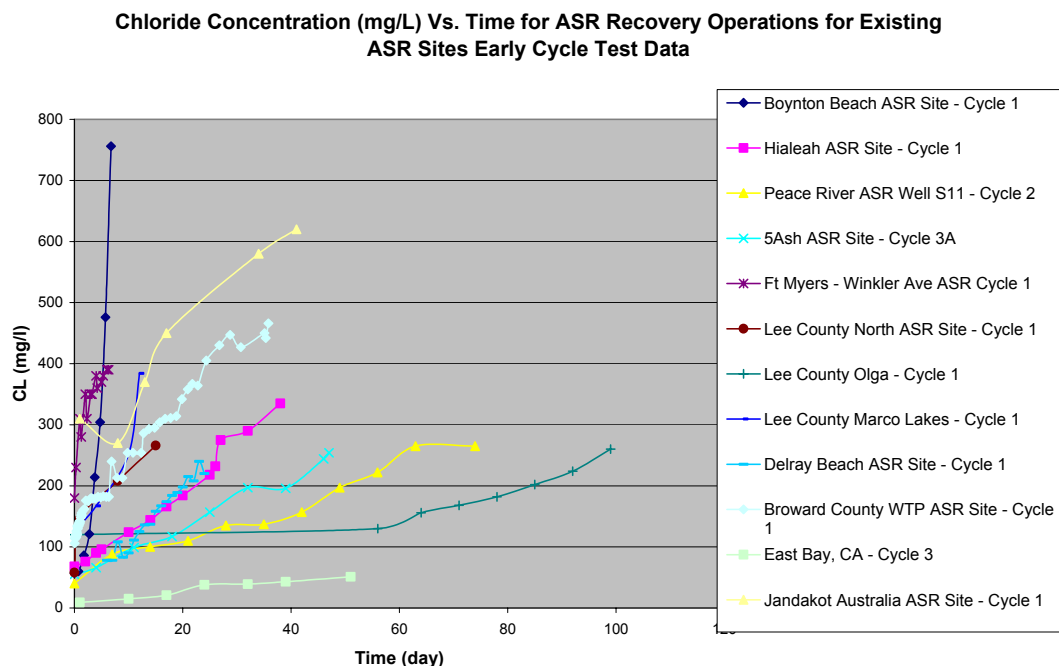


Figure 3-8. Time versus chloride concentration for select ASR sites discussed in this report.

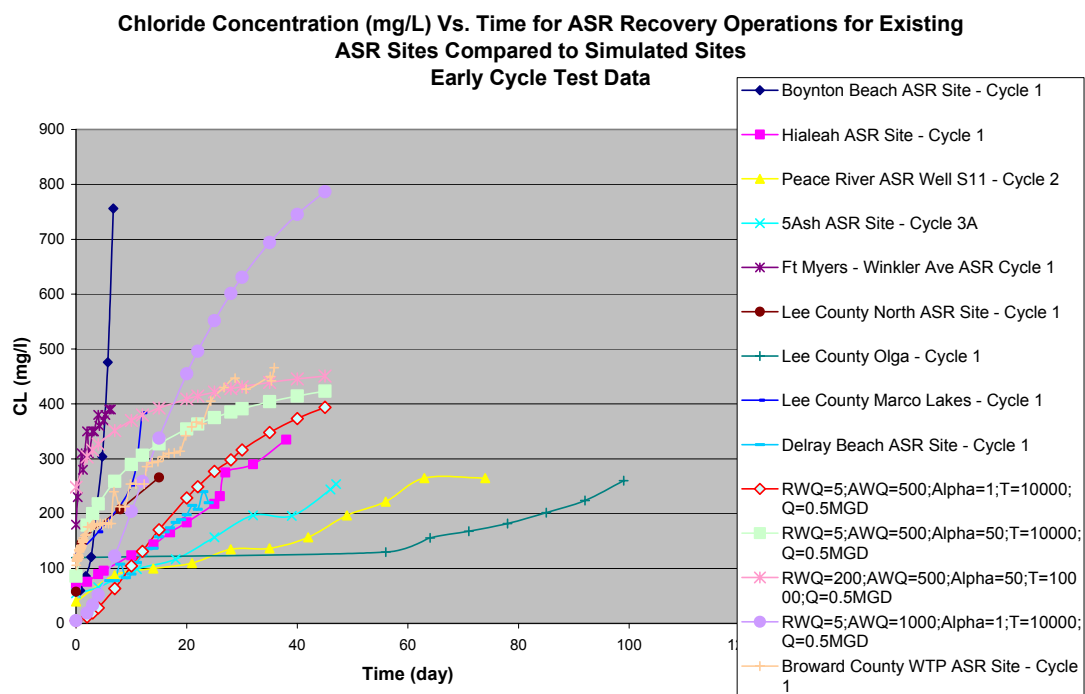


Figure 3-9. Time versus chloride concentration for select ASR sites discussed in this report and model simulation results.

The matrix of model simulations (e.g., the synthetic data set) included a wide range of values for each parameter of interest. A portion of the model run matrix with RE percentages for each is shown in Appendix B of this report. Once the synthetic data set was complete, tables of data were constructed in a spreadsheet to develop statistics and further evaluate the results for trends. In addition, some variables were further combined or transformed into logarithms. Appendix B results also depict these various combinations and logarithms. Once the data were sufficiently organized and reduced, the popular statistics program Statpro® was utilized to complete a multiple linear regression analysis. Each regression model tested variables at the 99% significance level. Multiple regression models were evaluated using various combinations of variables, transformed variables, or combination variables. In the end, the regression analysis indicated that the most important variables were RI, the logarithm of RI, the logarithm of the inverse relative dispersivity, dispersivity, the logarithm of dispersivity, and the recharge volume. Of the initial ten regression models evaluated, two provided the highest r^2 coefficients. Model 1 provided the best prediction of RE percentage as compared to the numerical model results. Model 5 also provided good predictions, although it used a slightly different combination of variables. Both models suggested that the proposed use of dimensionless parameters composed of the tested variables would be a useful approach in evaluating prospective ASR project performance. Model 1 simulation results are shown on figure 3-10 compared to numerical model results along with the model equation and r^2 coefficient. Similar model # 5 results are shown on figure 3-11.

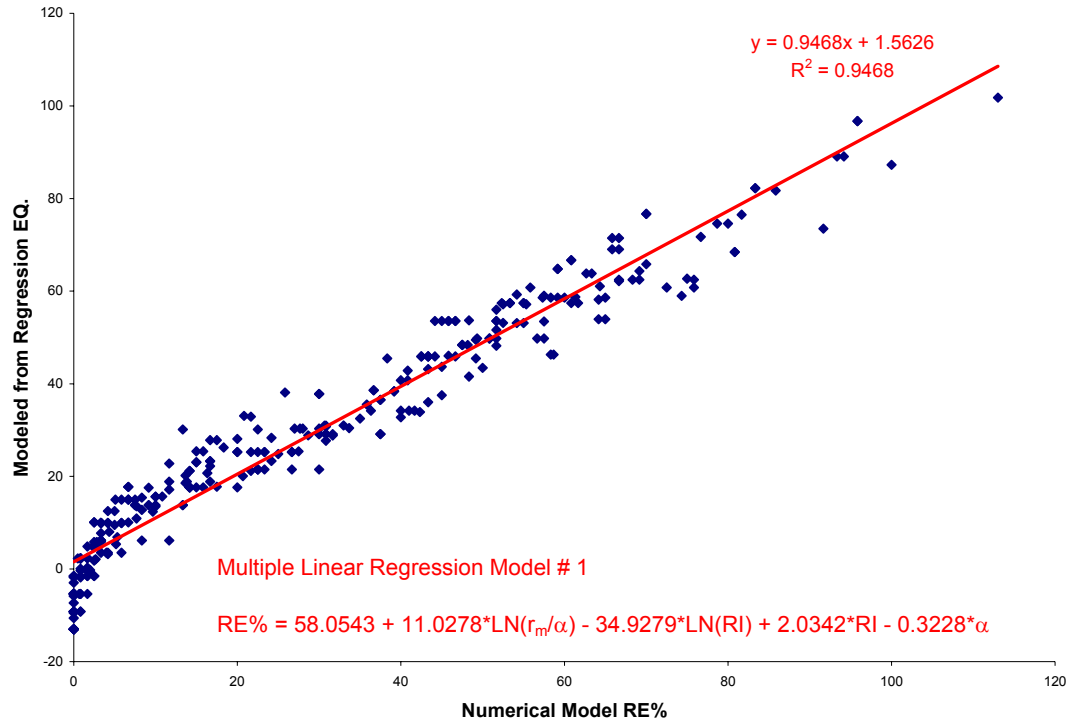


Figure 3-10. Multiple linear regression model # 1 results compared to original numerical model predictions of RE percentage.

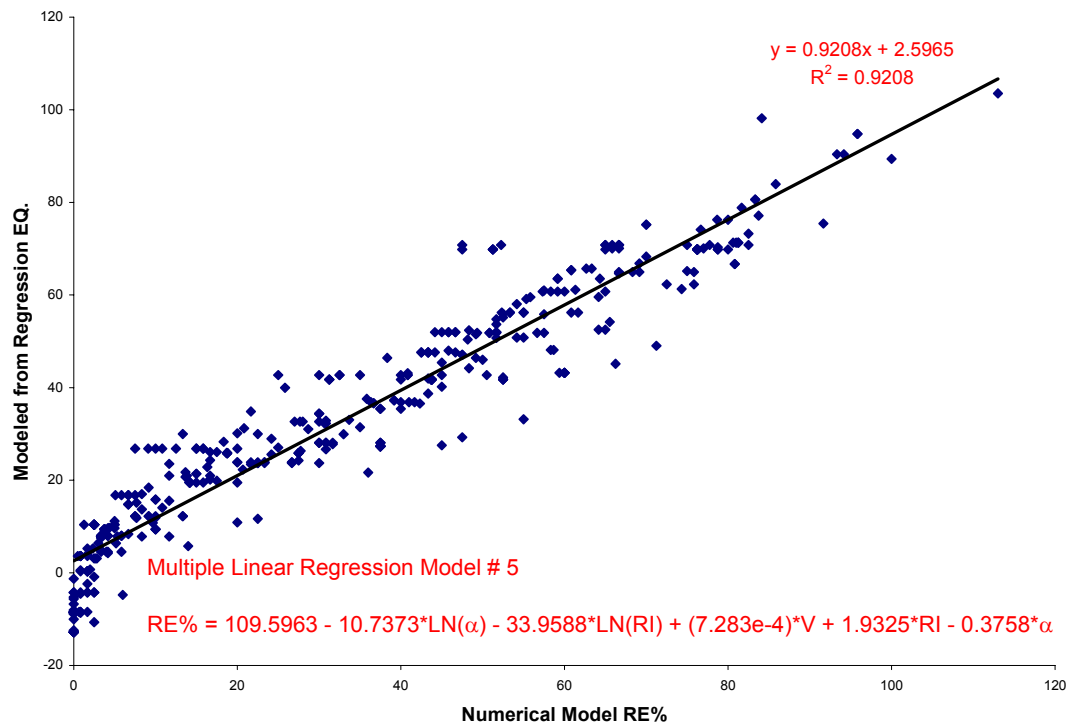


Figure 3-11. Multiple linear regression model # 5 results compared to original numerical model predictions of RE percentage.

The best model was reasoned to be model # 1 since it had the highest r^2 coefficient, however, it is important to note that the RI and logarithm of RI is common to both models. Also, if the last two components of model # 1 equation are removed so that the model only includes the two proposed dimensionless parameters, the inverse relative dispersivity and the recovery index, the r^2 is still 0.91. Therefore, the multiple linear regression modeling has confirmed the viability of using the two dimensionless parameters as ASR performance metrics to allow optimization of the site selection as well as the final design. In order to provide additional aid to the ASR practitioner, the numerical model results and the dimensionless parameters were utilized to develop theoretical ASR performance envelopes. The envelopes provide a graphical method to estimate the initial cycle one RE percentage provided that basic water quality and hydrogeologic data are available. The performance envelopes were also used to estimate near-field dispersivities at the existing brackish-water ASR project sites. The development of the envelopes and their potential uses are discussed in the next section of this report.

ASR brackish water performance envelopes

ASR brackish water performance envelopes were developed using the numerical model results discussed previously. As was discussed in previous sections of this report, the numerical model results were compiled, mathematically transformed, and combined for testing purposes using multiple regression analysis. The results of that testing confirmed that both proposed dimensionless parameters, the inverse relative dispersivity (R_{α}') and the recovery index (RI), are viable as ASR performance metrics. Obviously, the metrics could be utilized in the proposed multiple linear regression model # 1

directly, however, a graphical method may be more simplistic for potential users to grasp. Therefore, multiple graphical methods were compiled and reviewed as part of this research effort. Many of the graphical methods investigated were ultimately rejected due to their complexity or usefulness. In the end, a series of plots were constructed that relate the logarithm of the R_a' and the logarithm of RI to the initial cycle one RE percentage. Again, chloride was exploited as the water quality regulatory parameter imbedded within the RI. Initially, all of the numerical model data and the calculated dimensionless parameters were plotted on one single graph. This first iteration graph is shown below as figure 3-12. In order to incorporate the logarithm of RI into a two-dimensional graph, a family of different RI lines has been compiled onto a single graph. The graphs also reveal the r^2 coefficient for each set of RI values as compared to a perfect line.

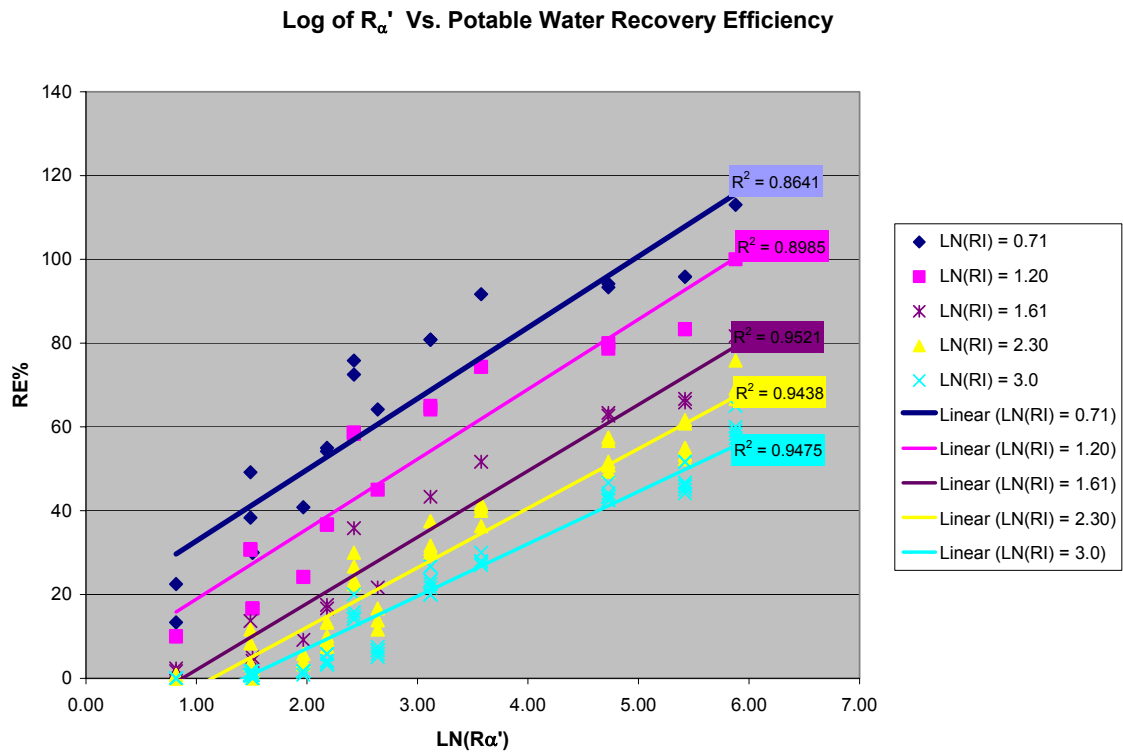


Figure 3-12. ASR RE percentage as a function of inverse relative dispersivity (R_a') and recovery index (RI). A “family” of RI graphs is shown to facilitate use.

As the RI values decrease, the r^2 coefficient declines from 0.9475 to 0.8641, indicating a weaker relationship at lower RI values. In addition, the RI data exhibit a wide spread in some cases such as logarithm RI equal to 2.30. A variation of figure 3-12 was tested that further segregated the data into similar “bins” of like ambient groundwater chloride concentrations while keeping the remaining parameters the same. Although this results in a series of five graphs rather than one, the effort was fruitful as depicted on figures 3-13 to 3-17. In addition to the theoretical ASR performance envelopes shown on the graph, real ASR field results from projects discussed in this report are also plotted. The real site data are plotted on the most relevant figure based upon ambient chloride concentration bin that best fits a particular site; therefore in cases where the AWQ was between bin thresholds, a site was plotted on more than one graph.

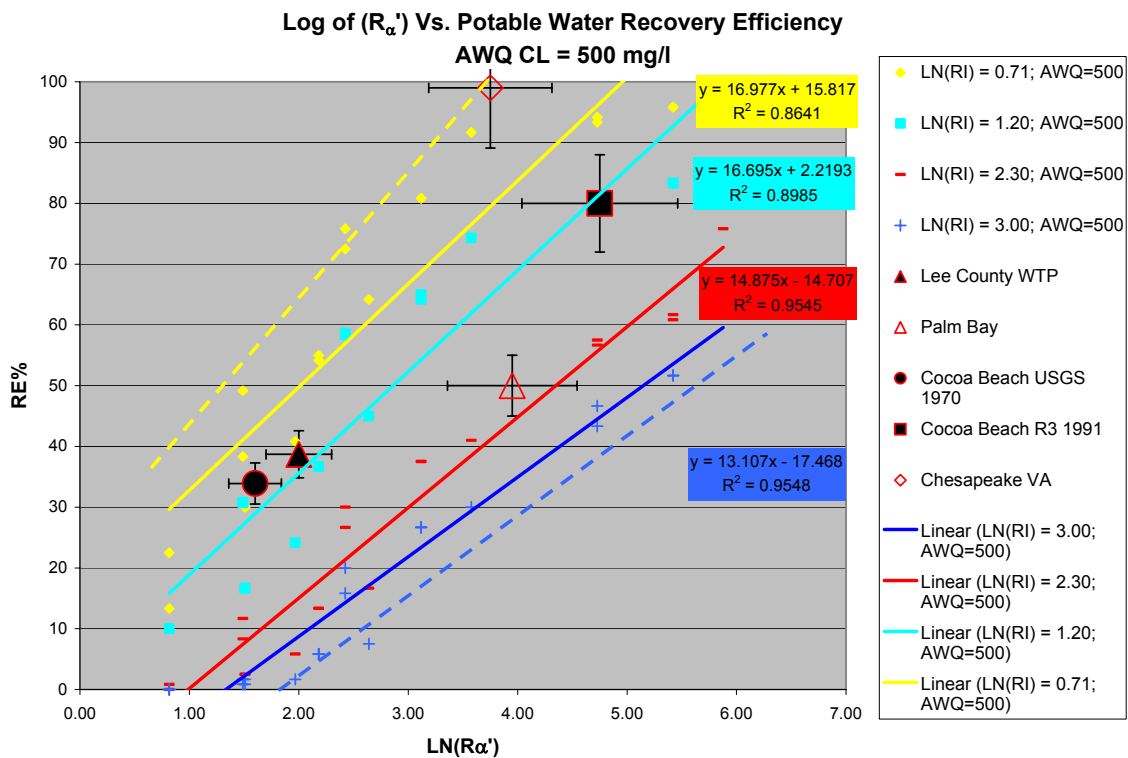


Figure 3-13. ASR RE percentage as a function of inverse relative dispersivity (R_{α}') and recovery index (RI) for AWQ equal to 500 mg/l.

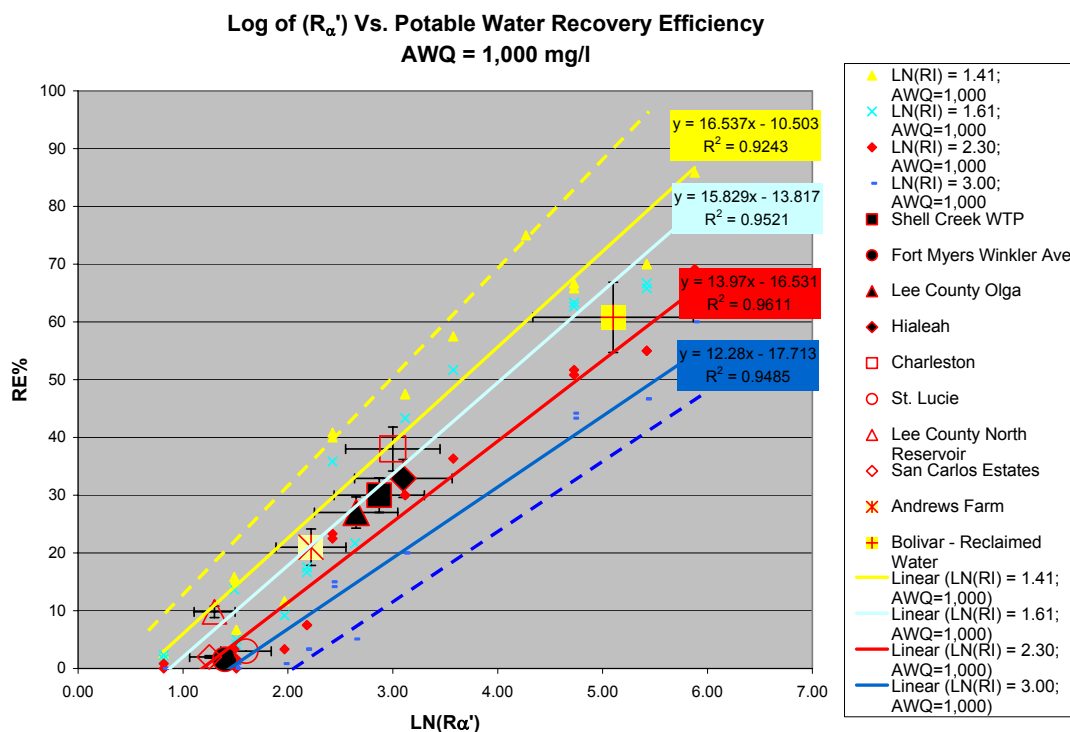


Figure 3-14. ASR RE percentage as a function of inverse relative dispersivity (R_{α}') and recovery index (RI) for AWQ equal to 1,000 mg/l.

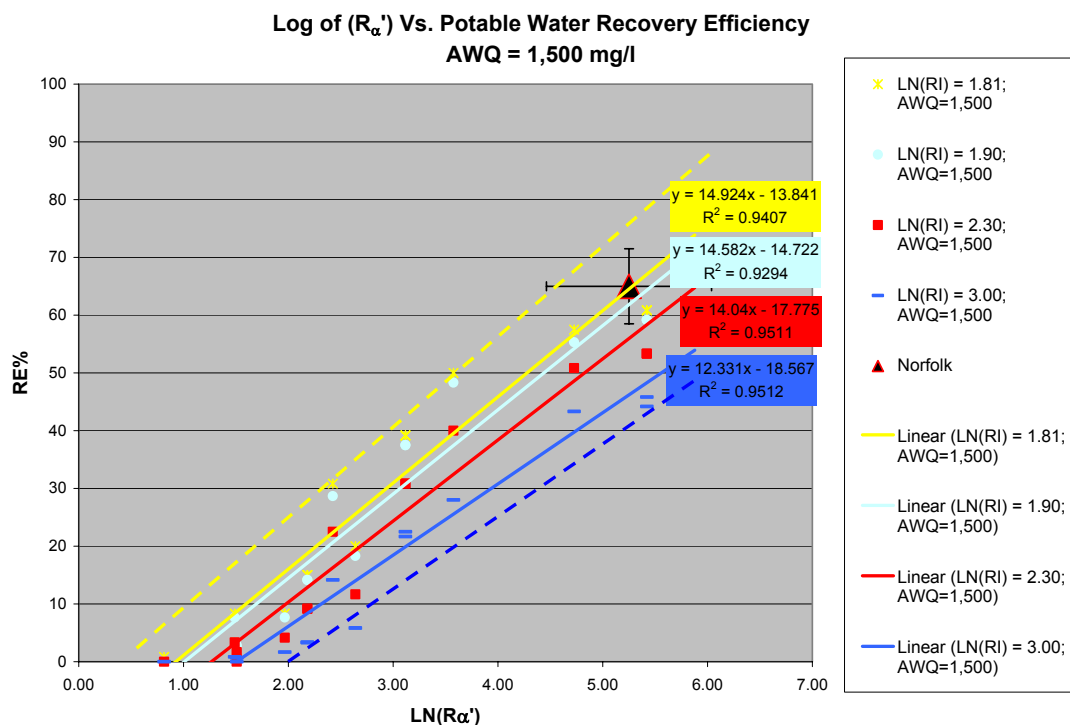


Figure 3-15. ASR RE percentage as a function of inverse relative dispersivity (R_{α}') and recovery index (RI) for AWQ equal to 1,500 mg/l.

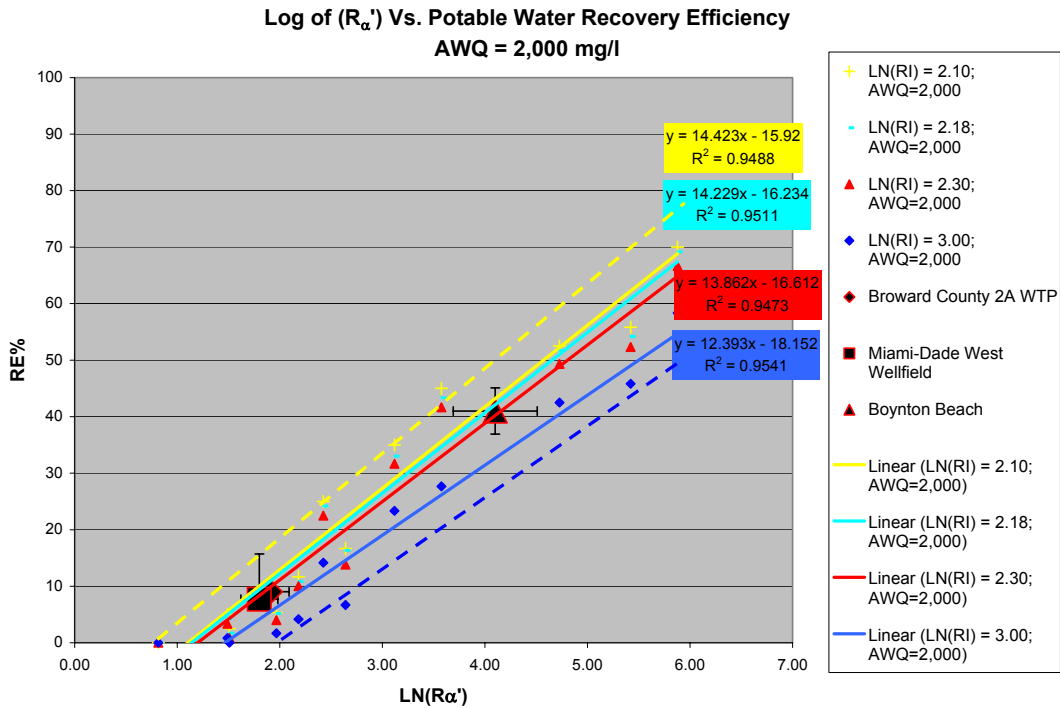


Figure 3-16. ASR RE percentage as a function of inverse relative dispersivity (R_{α}') and recovery index (RI) for AWQ equal to 2,000 mg/l.

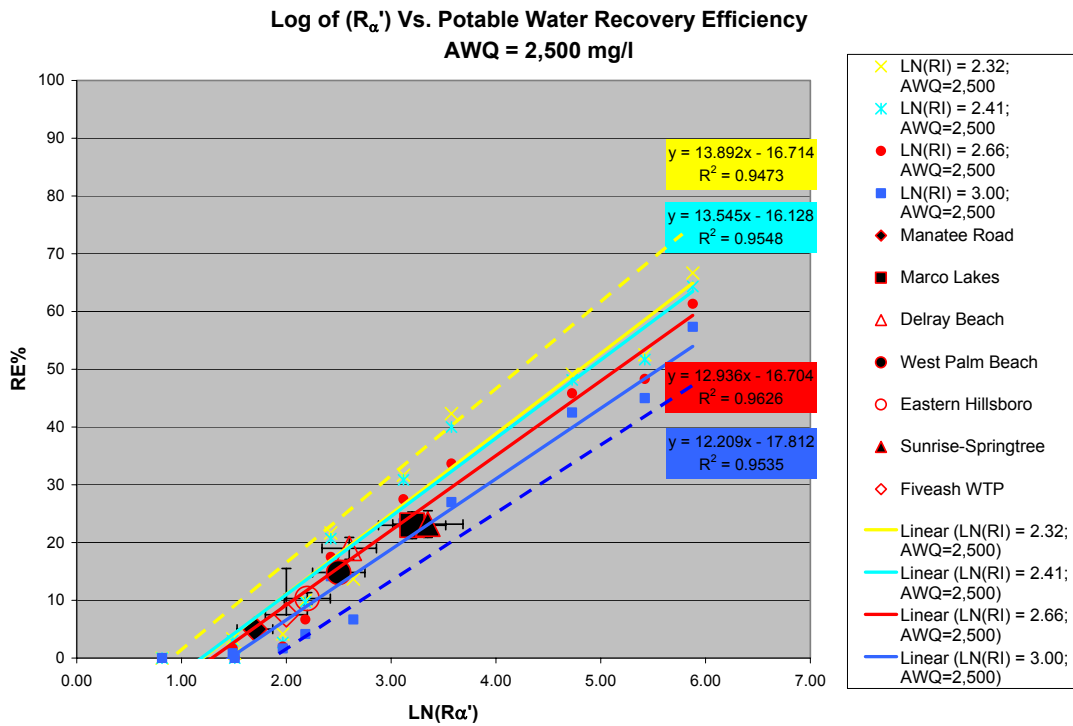


Figure 3-17. ASR RE percentage as a function of inverse relative dispersivity (R_{α}') and recovery index (RI) for AWQ equal to 2,500 mg/l.

The performance envelopes shown on the five figures are all similar except that the width of the envelope decreases with increasing ambient groundwater chloride concentration. The real field data from the ASR project sites supports this notion. For instance, at 500 mg/l of chloride in the ambient groundwater, the real field results vary considerably. At 2,500 mg/l of chloride in the ambient groundwater, the real field results plot close to each other in a rather narrow band. It is important to note that none of the field results plot outside the theoretical envelope derived from this research. Plotting the real field results required three key pieces of data, namely the ambient groundwater chloride concentration, the recharge water chloride concentration, and the RE percentage. Since there is error associated with each of those parameters, error bars are included with each field data value. For instance, at some of the sites, the ambient groundwater is stratified within the aquifer so that the chloride concentration worsens with depth. Also, the recharge chloride concentration varies throughout the recharge period; for a few sites the variation was considerable. The reported RE percentage itself is questionable in a few cases where the cycle one recovery was ended prior to exceeding the 250 mg/l regulatory standard. As was discussed and shown in Chapter 2, data curve fitting was utilized to estimate the full RE percentage for these sites. Lastly, for some sites, other water quality indicator parameters were utilized for monitoring purposes on a routine basis as compared to chloride. The Eastern Hillsboro ASR site is one such example where specific conductance was utilized for operational purposes due to its ease of use and low cost.

In order to utilize the ASR performance envelopes for ASR planning purposes, the user will need to estimate the average groundwater chloride concentration within the

storage zone, estimate the recharge chloride concentration, assume a recharge volume of water, and estimate a dispersivity based upon the aquifer geologic characteristics as has been discussed previously or utilize equation 6 presented previously. The porosity is assumed to be 25% for the performance envelopes. If actual field data are available at a prospective site, the site data should be utilized rather than assumptions or estimates.

Once the necessary data are compiled, the user would calculate the logarithm of RI and the logarithm of R_{α}' , and then estimate the cycle one RE percentage from the appropriate chart. For project sites where the ambient chloride concentration is significantly greater than 2,500 mg/l, another estimating approach is recommended since the charts cannot account for large density stratification effects. In cases where the porosity is thought to be extremely low, such as the case of dual-porosity aquifers or fractured hard-rock aquifers, the charts should be used for screening purposes only since they would most likely overestimate the actual RE percentage from such systems.

Cumulative recovery efficiency as a metric

As shown in Chapter 2 of this report, the Cumulative Recovery Efficiency (CRE) appears to be a valuable long-term ASR performance measure. The modeling completed for this report indicates that a best-fit logarithmic curve of CRE percentage versus number of cycles could be utilized to make preliminary estimates of performance over time. Certainly if a few test cycles have been completed, the best logarithmic relationship could be established from the testing results. In order to use the CRE as a feasibility tool, the ASR practitioner should probably develop a range of estimates based upon Figures 3-18 and 3-19.

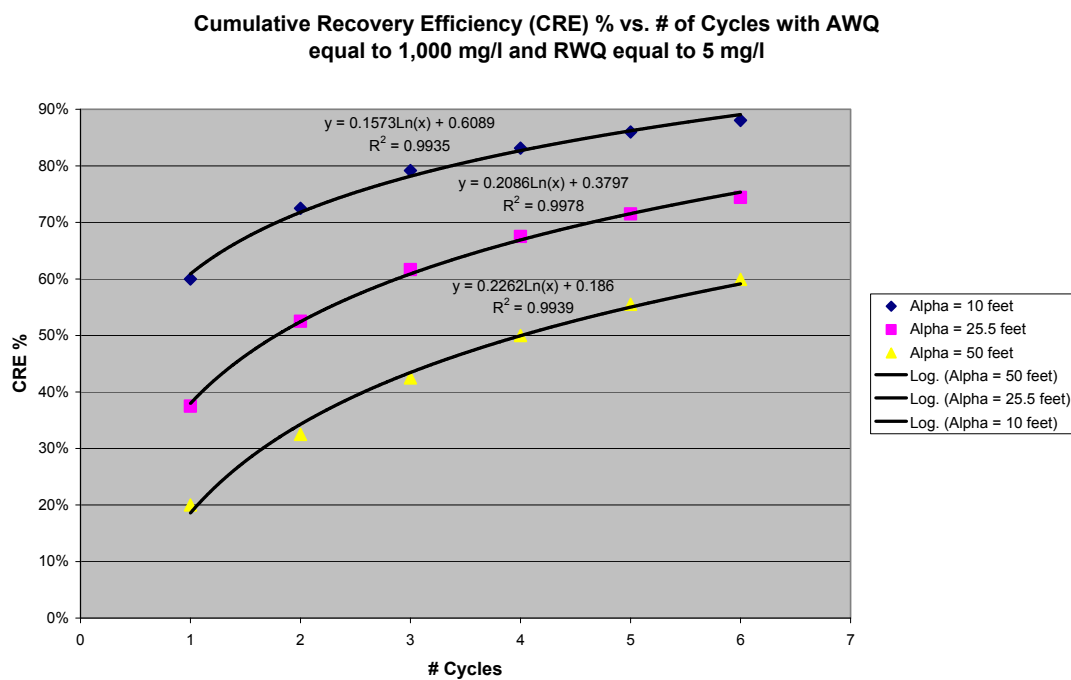


Figure 3-18. ASR CRE percentage versus # of recharge and recovery cycles for AWQ equal to 1,000 mg/l.

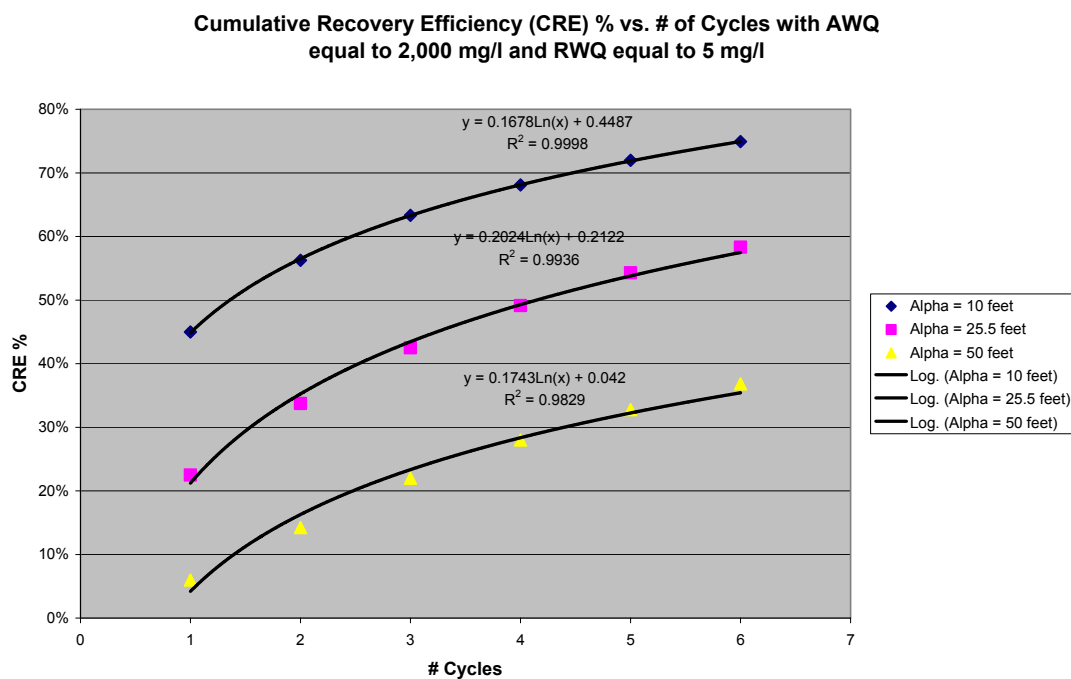


Figure 3-19. ASR CRE percentage versus # of recharge and recovery cycles for AWQ equal to 2,000 mg/l

For example, if the initial cycle 1 RE percentage has been estimated to be 15% and the AWQ is 2,000 mg/l, figure 3-19 could be entered to project the CRE after six cycles to be between 40 and 50%. It should be noted that the regression equations provide an estimate of CRE as a decimal from 0 to 1. This value needs to be adjusted to a percentage value.

Validation of new ASR performance metrics

Several multiple variable linear regression models have been developed as part of this research effort. The models provide a method to estimate ASR cycle one RE percentage as a function of key independent variables. Model # 1 has been shown to be the best model and is recommended for use. In order to test the model, it was validated against real ASR project data. At the real project sites, all information is generally known except for the dispersivity. However, as was discussed earlier in this chapter, dispersivity estimates have been estimated at a number of ASR sites. The estimates have been derived from field data or from numerical modeling. In addition, numerical modeling completed for this report has provided some additional values to use in validation tests. Table 3-28 provides a summary table with these values along with their reference.

In order to validate Model # 1 that was presented previously, the RE percentage was set equal to the observed value at each site above and then the “goal seek” function of Microsoft Excel® was utilized to find the associated dispersivity. The dispersivity estimate provided by the multiple regression model was then compared to results listed above. The results are presented on Table 3-29. Also, Figure 3-20 presents the real data versus the regression model data along with the resulting straight line if the model were perfect.

Table 3-28. ASR Sites with estimated dispersivity values

ASR Project Site	Predominant Geologic Character	Estimated Dispersivity (feet)	Reference
Charleston, USA	Limestone/sandstone	0.33	Pavelic et al. (2002)
Andrews Farm, Aus	Sandy limestone with conduits	82.03	Pavelic et al. (2002)
Broward County WTP 2A	Limestone	22.0	This report
Fort Myers Winkler Ave	Limestone	70.0	This report
Boynton Beach	Sandy Limestone	8.0	This report
Hialeah	Limestone	55.0	This report
Marco Lakes, USA	Limestone	30.0	M. Pearce
Lee County Olga, USA	Limestone	39.0	M. Pearce
Lee County North, USA	Limestone	78.0 considered poor estimated by M. Pearce	M. Pearce
Hialeah, USA	Limestone	65.0	Merritt (1997)

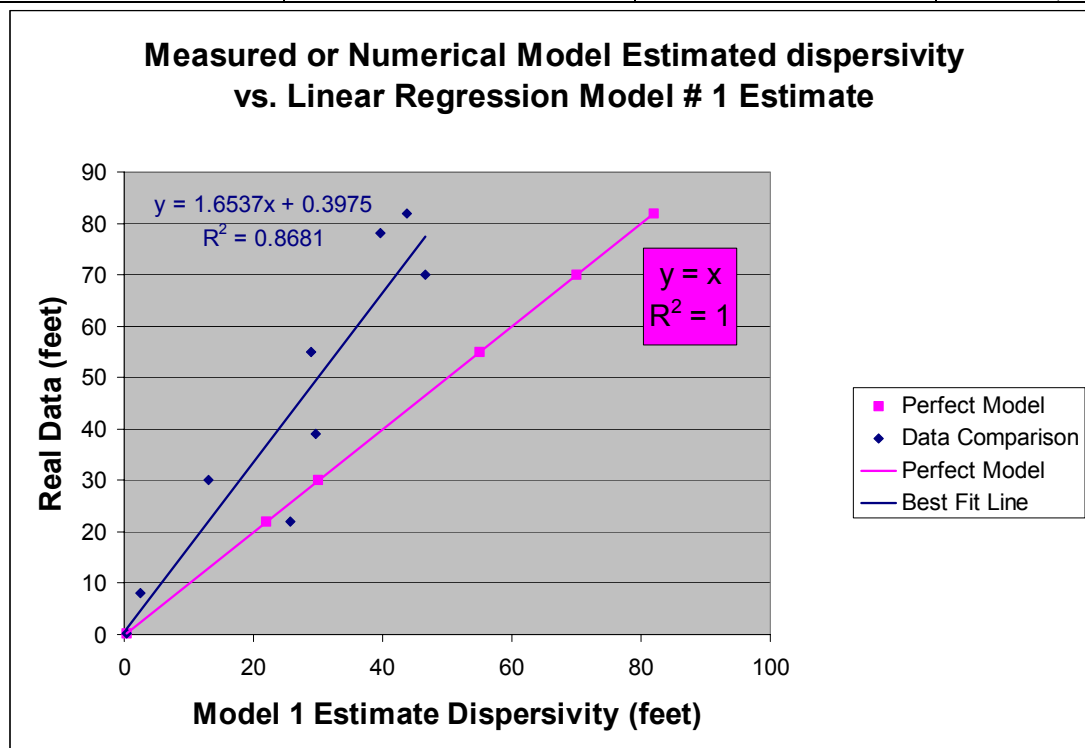


Figure 3-20. Real site dispersivity estimates versus linear regression model estimates. Also shown is the resulting straight line if the model were perfect.

Table 3-29. Real site estimates of dispersivity versus linear regression model estimates

ASR Project Site	Predominant Geologic Character	Real Dispersivity (feet)	Linear Regression estimate (feet)
Charleston, USA	Limestone/sandstone	0.33	0.41
Andrews Farm, Aus	Sandy limestone with conduits	82.03	43.77
Broward County WTP 2A	Limestone	22.0	25.79
Fort Myers Winkler Ave	Limestone	70.0	46.6
Boynton Beach	Sandy Limestone	8.0	2.58
Hialeah	Limestone	55.0	28.89
Marco Lakes, USA	Limestone	30.0	13.06
Lee County Olga, USA	Limestone	39.0	29.61
Lee County North, USA	Limestone	78.0 considered poor estimated by M. Pearce	39.56

In reviewing Figure 3-20, it is evident that the linear regression model underestimates the actual dispersivity. This bias was mentioned previously in this report and is thought to be due to two factors. First, the dispersivity tensor utilized in the 500 model simulations that the linear regression model is based upon was assigned the same value in three directions. For real ASR sites, the vertical dispersivity is probably far less than the lateral dispersivity. Second, the model simulations assumed that the porosity was 25% for all model runs. This value may be high for predominately limestone aquifers where values can range from 5 to 25%. Therefore, the real site data that represents mostly limestone geology, may actually have porosities closer to 15% on average. The resulting estimates provided by the multiple linear regression model are probably biased somewhat low. The correlation coefficient is still quite good at 0.87; therefore the model is probably okay as presented. In addition, a majority of the comparable “real” dispersivity estimates were

developed from contaminant transport modeling through calibration efforts. Typically, as commented on by Gelhar et al. (1992), these derived estimates are not as reliable as field derived values. Therefore, the model is recommended as presented with no major changes.

One weakness with the proposed regression model is that the ASR practitioner must select a reasonable estimate of dispersivity accounting for both the geologic character and the scale of the problem. For many ASR projects, the scale or r_m distance is less than 300 feet. Out of 27 brackish water ASR projects utilized and discussed in this report, only eight sites had an r_m distance of greater than 300 feet. Therefore, for a majority of planned ASR projects, the suggested dispersivity values presented in Tables 3-18 and 3-19 should suffice since scale effects are assumed to be less important than the geologic character. For larger ASR projects with large values of r_m , the approach presented by Schulze-Makuch (2005) is recommended with some variations. A key assumption of this approach is that the dispersivity is correlated to r_m , but is not totally dependent upon it. Both approaches were tested on the 27 brackish water ASR projects where all relevant data were available.

First, regression model # 1 was utilized with suggested dispersivity values segregated by geologic character as presented in Tables 3-18 and 3-19. Again this approach assumes that project scale is less important than geologic character. This approach provided RE estimates that were a close match to observed values. Figure 3-21 details the comparison of modeled versus observed RE values for 27 ASR sites. The figure reveals that the r^2 coefficient is 0.72 for this planning approach.

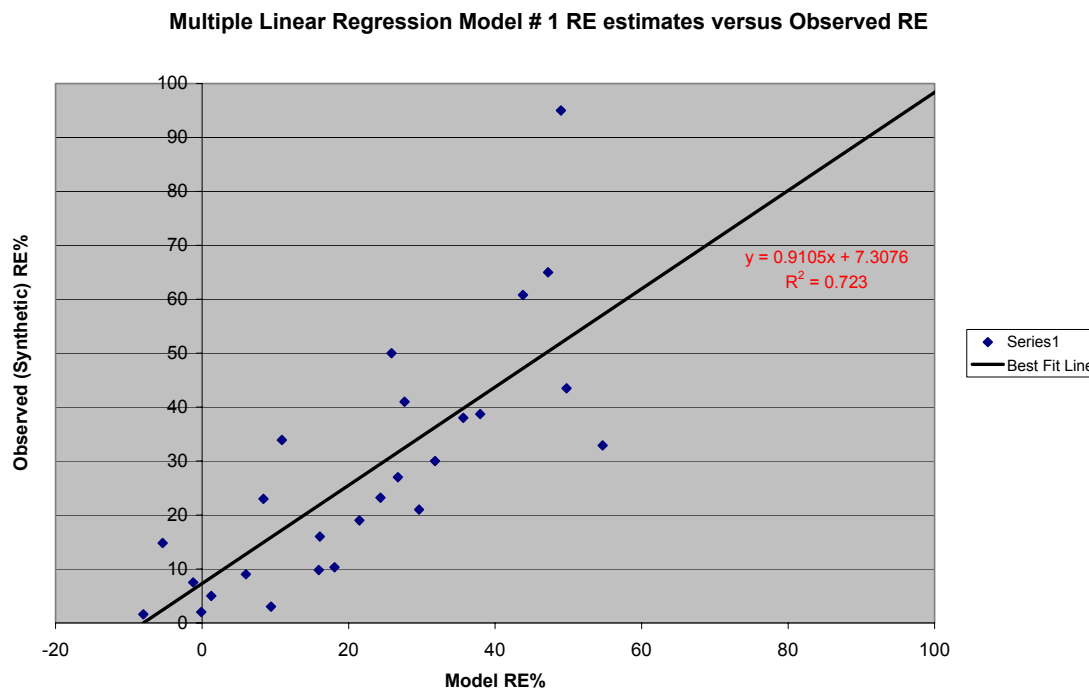


Figure 3-21. Real site RE estimates versus linear regression model RE estimates. Also shown is the resulting r^2 coefficient.

The second approach tested was estimating the dispersivity value for input into model # 1 via the analytical models proposed by Schulze-Makuch (2005). This approach assumes that dispersivity is both dependent on geologic character and correlated to project scale. These models are all “power law” models relating the dispersivity to both the geologic character (c) and to the scale of the project (L) using a scaling exponent (m). Equation 6 presented previously details the relationship. In the original work, Schulze-Makuch developed a proposed table of constants for c and m for unconsolidated aquifers, sandstones, carbonates, basalts, and granites. For the unconsolidated aquifer values, several constants were developed based upon the assumed reliability of the data. Table 3-30 summarizes the coefficients and exponents recommended for general use.

Table 3-30. Recommended power law coefficients and exponents for estimating dispersivity values.

Type of Medium	Coefficient c	Scaling exponent m
Unconsolidated – using two most reliable datasets	0.112	0.67
Sandstones	0.01	0.88
Carbonates	0.80	0.74
Basalts	0.15	0.75
Granites	0.21	0.53

The first attempt to utilize these values to provide dispersivity estimates for import into regression model # 1 produced a poor RE estimate as compared to the observed values. In reviewing the basis for the recommended c and m values for carbonates, it became clear that the carbonate medium should be sub-divided. For the carbonate ASR projects in south Florida and Australia, the carbonates can be sub-divided into three general categories, namely, sandy limestone, vuggy limestone with minimal conduits, and karst limestone with conduits. The c and m values presented for carbonates in Table 3-30 are mostly based upon karst limestone sites as determined from the original research summaries. Therefore, for this study, two additional medium categories are proposed including sandy limestone and vuggy limestone. This approach is consistent with previous discussions in this report emphasizing a “continuum approach” for the aquifer geologic character. Sandy limestone may resemble a sandstone more than a karst limestone, therefore, in a continuum approach, the c and m values should be much more similar to the recommended sandstone values. The vuggy limestone is thought to be an average of the sandstone and karst limestone values. These ideas were incorporated into the original Table 3-30 in order to produce a revised list of model coefficients. Table 3-31 provides an amended list of coefficients including the newly proposed categories.

Table 3-31. Recommended power law coefficients and exponents for estimating dispersivity values including new medium categories developed for this research report.

Type of Medium	Coefficient c	Scaling exponent m
Unconsolidated – using two most reliable datasets	0.112	0.67
Sandstones	0.01	0.88
Sandy Limestone	0.208	0.845
Vuggy Limestone	0.405	0.81
Karst Limestone	0.80	0.74
Carbonates	0.80	0.74
Basalts	0.15	0.75
Granites	0.21	0.53

These amended values were then utilized to develop dispersivity estimates for input into regression model # 1. The model estimates of RE were much closer to the observed values of RE using the amended medium coefficients and exponents presented in Table 3-31. Figure 3-22 presents the estimated RE values versus the observed RE values for this planning approach along with the r^2 coefficient. Although the correlation coefficient is only 0.58, this planning approach appears to have some utility for those ASR projects where scale as measured by r_m is much larger than 300 feet.

Proposed New Planning Decision Framework

ASR Suitability Index

ASR site selection efforts could be greatly simplified through the application of a simple “suitability index”. This idea would aid the ASR community of practice enabling more simplistic and efficient site selection studies to be completed. The suitability index would be a spatially dependent measure of potential ASR feasibility. The various site selection criteria can vary depending upon the planning factors determined to be important for each site. Ultimately, the site selection criteria should be normalized into an ASR site selection suitability index from 0 to 1.

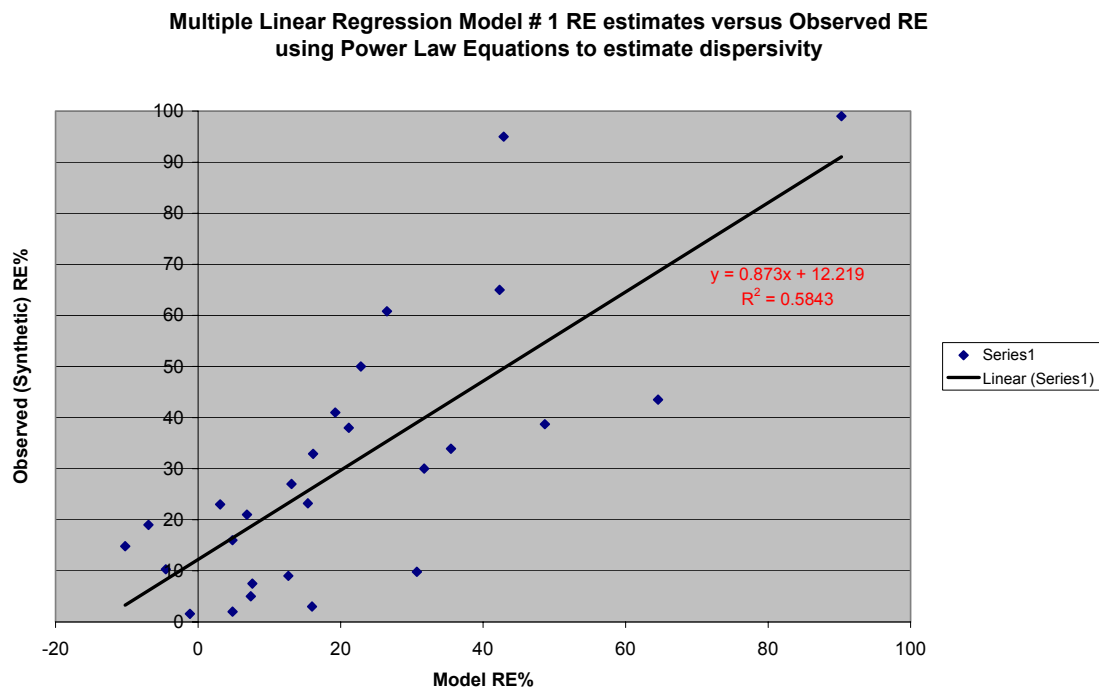


Figure 3-22. Real site RE estimates versus linear regression model RE estimates using Power Law equations to estimate dispersivity. Also shown is the resulting r^2 coefficient.

The USACE has traditionally utilized site selection evaluations to minimize impacts to environmental systems through application of water resources planning principles promulgated in 1983 (U.S. Water Resources Council, 1983). Water resources projects, including development of flood control and navigation projects, sought to maximize national economic development while minimizing impacts to the “environmental quality account” of each project. Typically, a series of overlap maps were prepared that presented planning factors like topography, soils, hydrology, location of threatened or endangered species, or location of other affected populations. Overlay methodology has been utilized by many Federal and State agencies (FDOT, 1999) and is discussed in more detail by Focazio et al., (2002). Overlay methods have been shown to be well suited for feasibility studies evaluating artificial recharge.

With the advent of Geographic Information Systems (GIS), more sophisticated overlay evaluations are now possible. Shahid (2000) discusses the use of overlay methodology in combination with remote sensing and GIS to evaluate vulnerability of groundwater aquifers to pollution. A similar approach has been used by other researchers.

Combining various GIS coverages into normalized site selection indices has been a recent technical advance. Tegelmark (1998) combined factors such as regional climate, topography, soil properties, and vegetation into a suitability index to predict natural Scots pine forest regeneration. Tegelmark used multivariate regression models in combination with overlay maps to find most suitable regions for reforestation. Xinhai et al. (2002), used GIS to combine information on topography, vegetation, rivers, roads, and location of villages/towns in order to develop a suitability index for crested ibis habitat. The habitat suitability index was normalized between values of 0 and 1. An integrated map of ibis habitat quality was prepared and compared to actual distribution of ibis in regions of China. Tseng et al. (2001) combined five GIS themes into a site selection index for locating optimal artificial reef sites. A decision support system was also used to supplement the GIS themes and ensure objective ranking of various criteria.

The performance of an ASR system in this environment is a complex process that may be affected by operational design, subsurface heterogeneity, density-dependent flow processes, and biogeochemical processes. One constraint to ASR implementation is aquifer zones with inadequate transmissivity that would not be able to accommodate large storage volumes due to unsustainably high induced aquifer pressures. In addition, highly heterogeneous aquifer zones may lead to enhanced mixing and hydrodynamic

dispersion (Merritt, 1986; Anderson & Lowry, 2004). Cavernous or “karst” zones within the ASR storage zone may be especially problematic. Groundwater with high concentrations of total dissolved solids (TDS) may lead to poor quality water being recovered by the ASR well (Pavelic et al., 2002), thereby limiting overall recoverability. In addition, high TDS values may cause buoyancy stratification of stored water due to density differentials between the ambient brackish groundwater and the recharged freshwater (Missimer et al., 2002). In addition to hydrogeological related performance factors, other important ASR site selection factors include:

- Availability of source water for recharge
- Quality of the source water
- Distance from the source water to the ASR well
- Landuse and availability
- Access constraints (e.g., roads for access and construction)
- Location of ecologically valuable habitats, including endangered species
- Location of system demands
- Location of existing groundwater users
- Availability of power
- Operational flexibility

The actual site selection criteria to be utilized at a potential project site should be customized to ensure that all local selection criteria are incorporated, however, criteria discussed in this report should be considered since they have been compiled from existing literature and reports. An example of an ASR suitability index is available in Brown et al. (2005b). In this article, Brown et al. utilize eight different site selection criteria to locate optimal ASR project sites in support of the Everglades ASR project.

The eight criteria that were utilized are:

- Ecological Suitability – Based on rarity, sensitivity, and/or value of habitat for plants, fish, and wildlife, and likelihood of presence of Federally threatened or endangered species (Rank 0 to 2; 2 for high suitability, 1 for medium suitability, and 0 for low suitability)

- Use and Density Factor – Known Floridan Aquifer System well users (Rank 0 to 2; 2 if no wells are located in a polygon, 1 if 1 to 5 wells within polygon, and 0 if more than 5 wells within polygon)
- Water Quality Assessment – Based upon published characterization of source water quality under the FDEP 305b clean water program (Rank 0 to 2; Rank 2 if source water fully meets standards, 1 if source water partially meets standards, and 0 if source water does not meet standards)
- Groundwater Quality of UFAS (Rank 0 to 2; Rank 0 if chloride concentration of the upper UFAS groundwater is less than 250 mg/l or greater than 3,000 mg/l., rank 1 if chloride concentration of UFAS is between 1,500 and 3,000 mg/l, and rank 2 if chloride concentration of UFAS is between 250 and 1,500 mg/l.
- Road Density (Construction/O&M Access) (Rank 0 to 2; Rank 0 if density of roads are low, 1 if density of roads are medium, and 2 if density is high)
- Locate near existing power lines (Rank 0 to 2; Rank 0 if not adjacent to existing power lines, 1 if near small KVA lines (in mostly urban areas), 2 if near major transmission lines)
- Pressure induced changes or high dispersive mixing potential, based upon aquifer transmissivity (Rank 0 to 2; Rank 0 if $T < 5,000 \text{ ft}^2/\text{day}$ or if $T > 25,000 \text{ ft}^2/\text{day}$, 2 if T is between 5,000 to 25,000 ft^2/day)
- Operational Flexibility (Rank 1 to 2; Rank 2 if close to Lake Okeechobee and major canal or major canal and CERP Impoundment or STA from Everglades Construction Project; Rank 1 for the rest)

Planning Decision Framework Procedure

In order to develop a new planning decision framework, ASR planning factors, planning guidance, and quantitative metrics have been compiled or developed. Guideline values and metrics are founded upon literature information, ASR case histories, and on simulation modeling using four separate ASR simulators. For the ASR case histories, real practical information and planning approaches were discerned from 50 ASR projects located around the world. Twenty of the sites were non-brackish water ASR projects while the remaining 30 sites were brackish water ASR sites. The important planning factors and considerations have been discussed throughout this report and will serve as a

basis for several ASR decision trees that provide the practitioner with a standardized procedure to follow when evaluating prospective ASR projects. The decision trees really form the skin of the ASR planning decision framework, while the supporting information, guideline values, and metrics form the internal organs and skeleton. As was demonstrated in the previous section, combining some of the site selection or evaluation criteria into a suitability index is an efficient means of selecting the best sites for further evaluation. Once the best sites have been selected, the new proposed ASR decision framework would then be utilized.

For each ASR project under consideration, a combination of different planning factors will determine its ultimate feasibility. The most important planning factors will vary across sites under consideration but all will have some commonalities. The decision framework is structured so that general ASR characteristics are evaluated first, followed by more specific requirements for brackish water ASR sites (if applicable). The general characteristics are important considerations for any ASR project, while the brackish water considerations are important considerations in only some potential projects. The framework relies upon guidance values and metrics developed as part of this report in combination with analytical or numerical models. The framework will provide a standardized methodology for evaluating new ASR projects. The framework is intended to be comprehensive and is integrated across the various ASR engineering and scientific planning factors and constraints.

As discussed in Chapter 1 of this report, the Federal water resources planning guidelines could serve as the starting point for developing the ASR decision framework.

According to the U.S. Water Resources Council (1983), the basic planning steps should consist of the following steps:

- Specification of the water and related land resources problems and opportunities associated with the Federal objective and specific State and local concerns.
- Inventory, forecast, and analysis of water and related land resource conditions within the planning area relevant to the identified problems and opportunities.
- Formulation of alternative plans.
- Evaluation of the effects of the alternative plans.
- Comparison of alternative plans.
- Selection of a recommended plan based upon the comparison of alternative plans.

For the ASR planning decision framework, some modifications of the six step planning process are required. For this research report the following twelve comprehensive steps are proposed to take a potential ASR water resources project from planning to construction to monitoring:

- Specification of the water and related land resources problems and opportunities associated with the project objective and specific local concerns.
- Inventory, forecast, and analysis of water and related land resource conditions within the planning area relevant to the identified problems and opportunities.
- Formulation of alternative plans.
- Evaluation of the effects of the alternative plans.
- Comparison of alternative plans including integrated risk and uncertainty analysis.
- Selection of a recommended feasible project based upon the comparison of alternative plans.
- Formulate pilot project plan
- Conduct pilot project plan
- Review results of pilot project plan
- Recommend full-scale project, proceed to next alternative, or abandon project
- Construct and operate full-scale plan
- Monitor and optimize full-scale plan

The proposed twelve step planning procedure is presented as a general overview flowchart in figure 3-23. Figures 3-24 to 3-28 provide more specific detail for the most important planning steps for the decision framework. Steps 6 and 10 are major decision

points in the process. One advantage to this approach is that some projects may not warrant advancement beyond step 6. This more thorough feasibility study, supported by

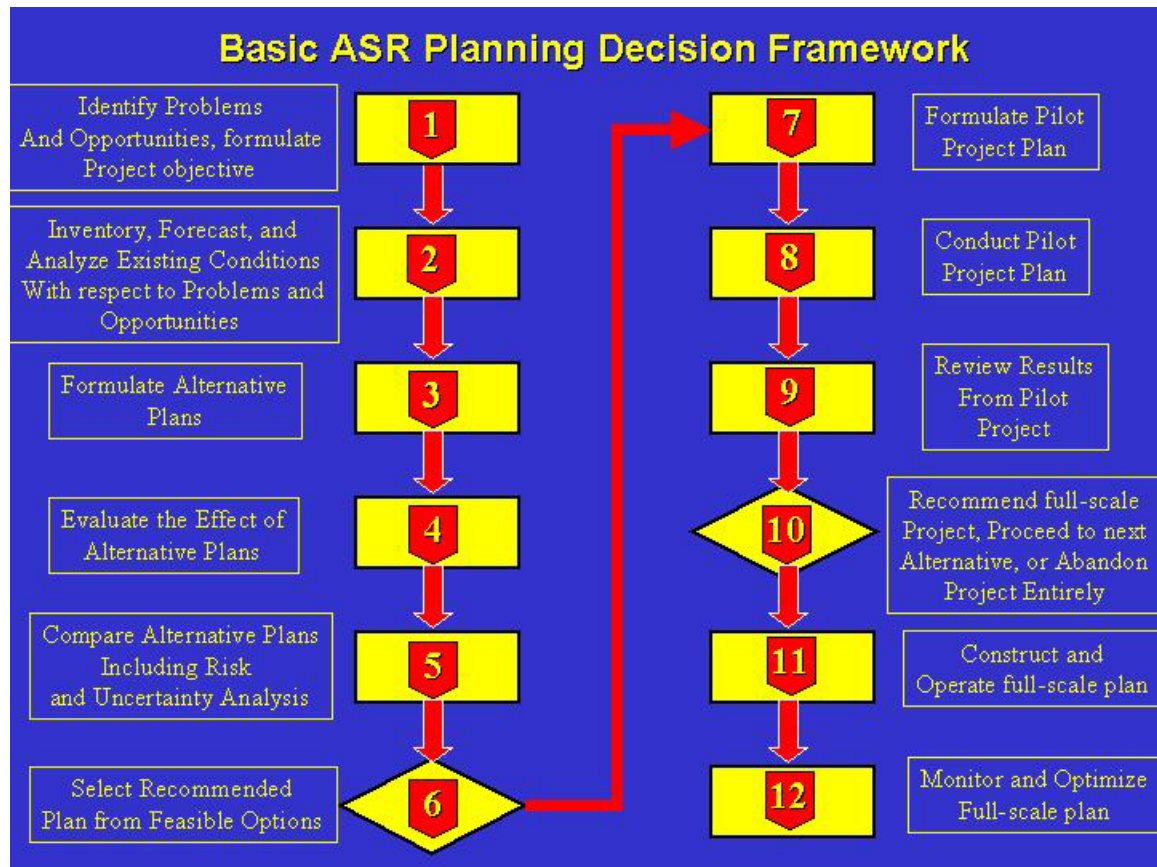


Figure 3-23. Proposed ASR Planning Decision Framework in generalized form.

more quantitative performance metrics and appropriate guideline values, may result in fewer projects going forward into pilot project stage, thus saving millions of dollars that would have been spent on exploration and well construction.

Figure 3-24 provides a more detailed overview of step 2 of the proposed planning decision framework. In step 2 the ASR planner gathers together key site inventory data in order to better evaluate the existing conditions in the project area. Key tasks in this step are the development of surface water availability and demand estimates. As pointed out previously in this report, without available water, AR or ASR projects are not

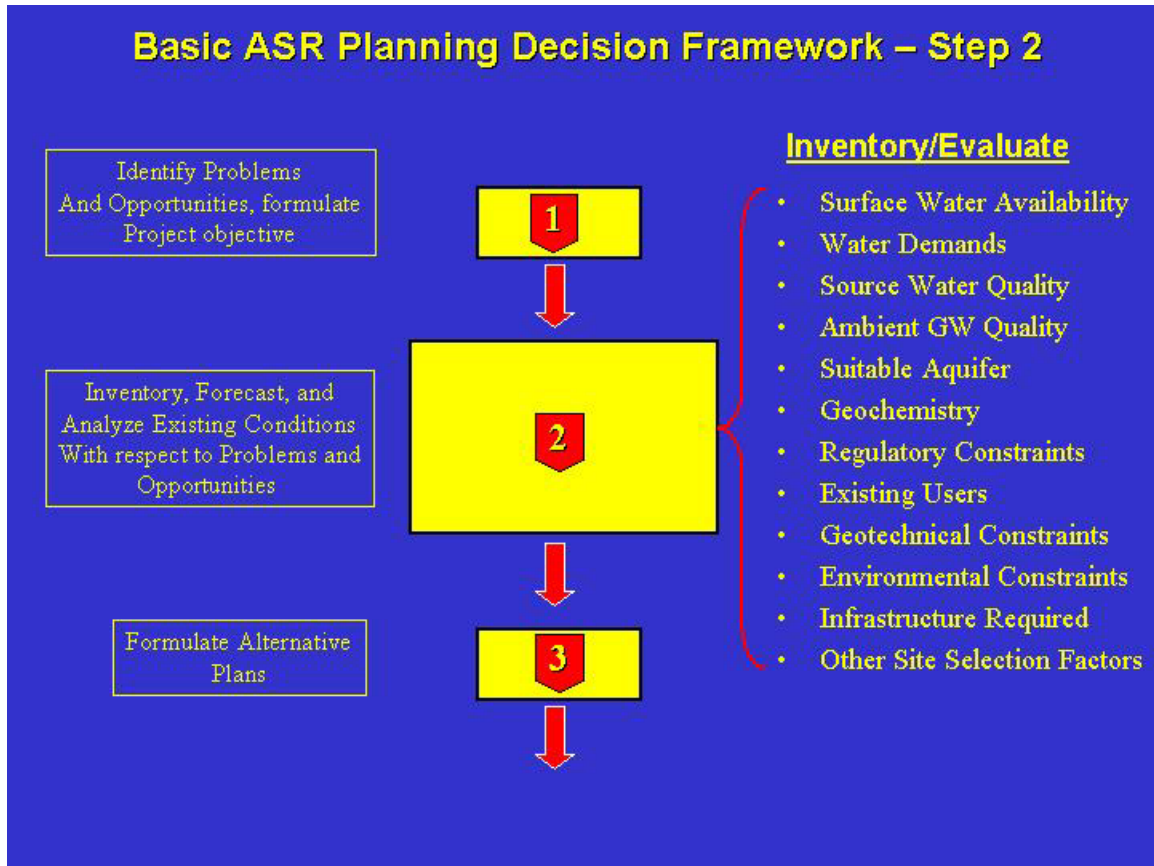


Figure 3-24. Proposed ASR Planning Decision Framework Step # 2 with key details of evaluations required.

feasible. Another key task is to determine if the proposed aquifer is suitable for large-volume ASR storage. Other site selection factors (e.g., power availability, landuse restrictions, location of users) should also be evaluated for each prospective site. The use of a “suitability index” may automate this task as discussed in the preceding section of this report. The suitability index may allow further automation of the ASR site selection process. The ASR planner should also develop a complete inventory of existing surface water and groundwater users in order to estimate baseline water needs. Users should include potable, irrigation, industrial cooling, and ecological categories to ensure that environmental impacts to be determined in Step 4, are all accounted for in the inventory.

Figure 3-25 provides a more detailed view of Step 3 where alternative plans are formulated. Certainly the most important issue to be evaluated during plan formulation is

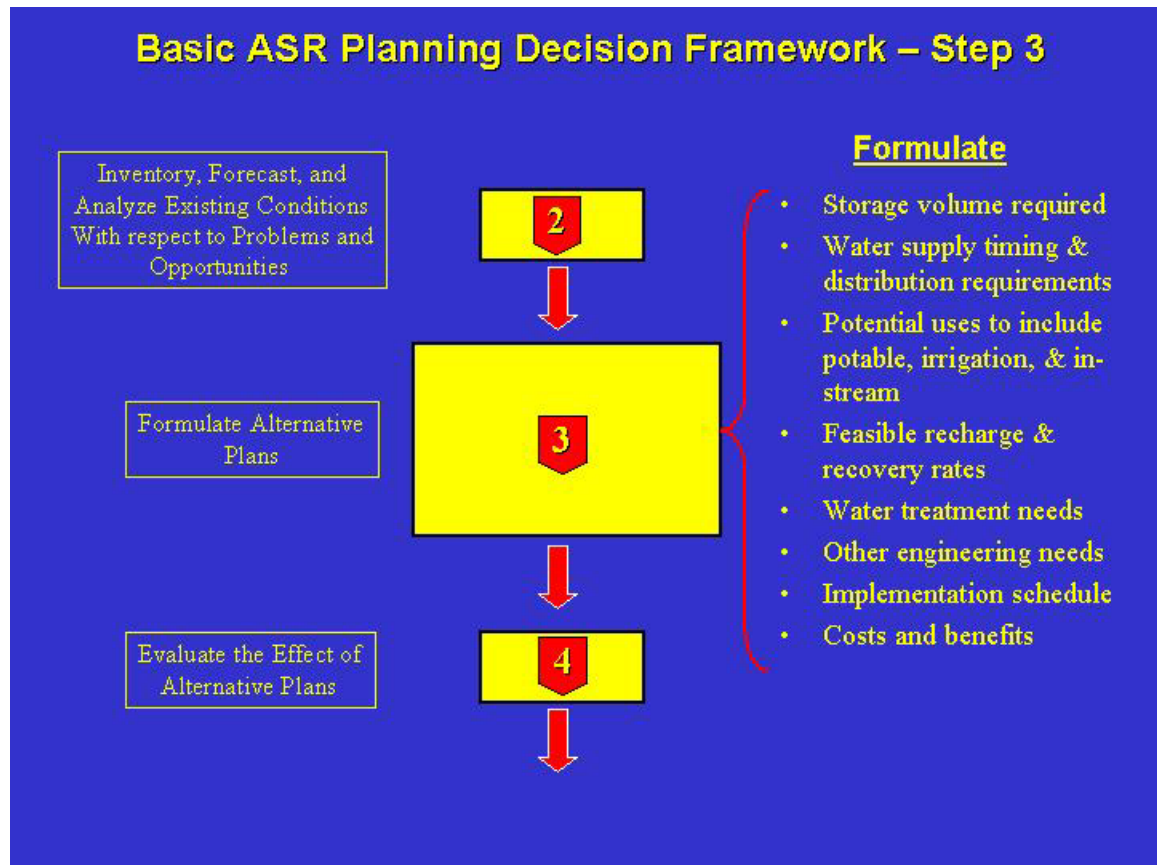


Figure 3-25. Proposed ASR Planning Decision Framework Step # 3 with key details of evaluations required.

the volume of water planned for storage and recovery. These overarching plan goals drive the remaining alternative evaluation process. Once the recharge and recovery volumes are known, various recharge and recovery rates can be evaluated and tested using analytical models discussed earlier in this report. Besides the recovery volume required for the alternative, the recharge water quality required has important planning ramifications. As was discussed previously, in-stream alternatives may require significantly better water quality than irrigation alternatives. The water quality needs in

turn drive the water treatment needs for each alternative. Once all of the key engineering requirements are enumerated, the preliminary costs and benefits can be estimated for

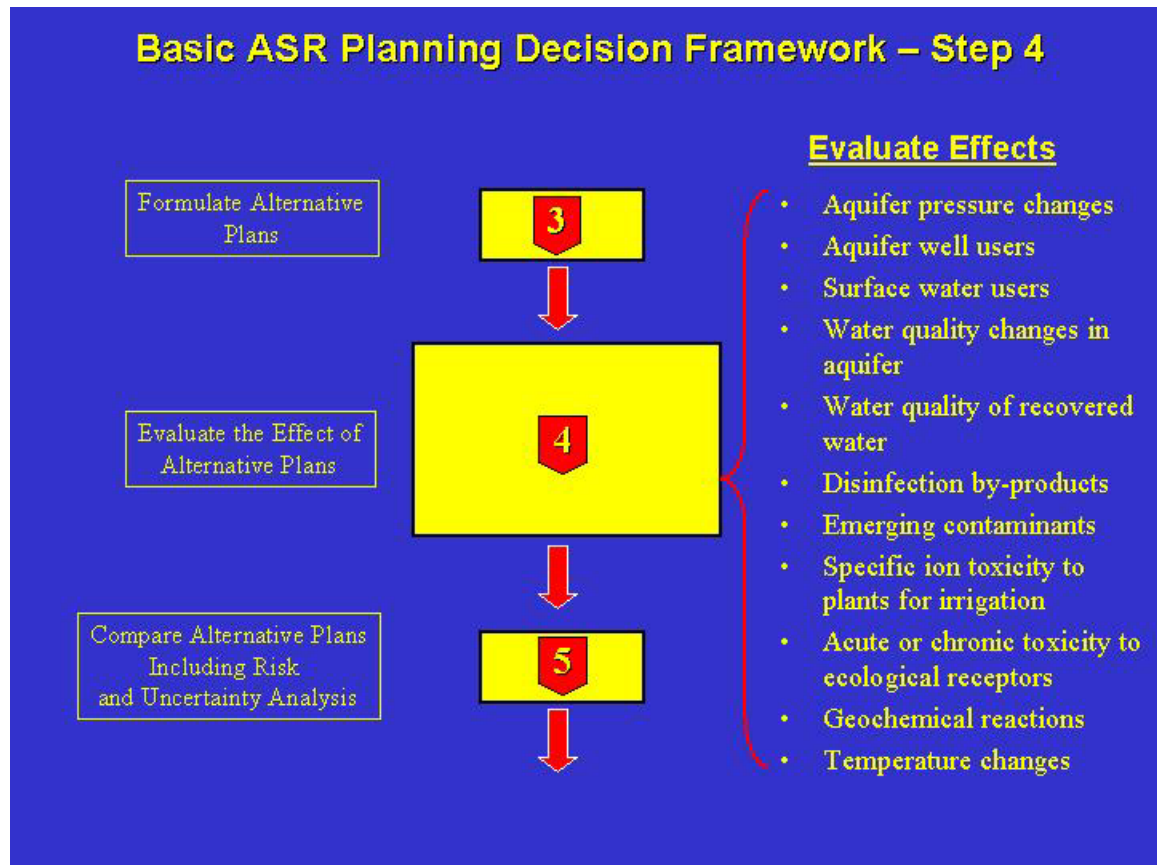


Figure 3-26. Proposed ASR Planning Decision Framework Step # 4 with key details of evaluations required.

each alternative. Figure 3-26 provides a detailed list of tasks necessary to complete the evaluation of effects for the various alternatives. As illustrated clearly in the figure, key evaluations are related to water quality, aquifer pressures, and direct effect of existing water users.

Figure 3-27 provides a detailed list of tasks necessary to complete framework Steps 5 and 6. Step 5 proceeds with a comparison of all of the alternate plans. Comparisons include expected recovery efficiency, estimated cumulative recovery, costs, benefits,

degree of environmental impact, and overall effect on users. An integrated risk and uncertainty analysis is also recommended in this step to ensure that each alternative is

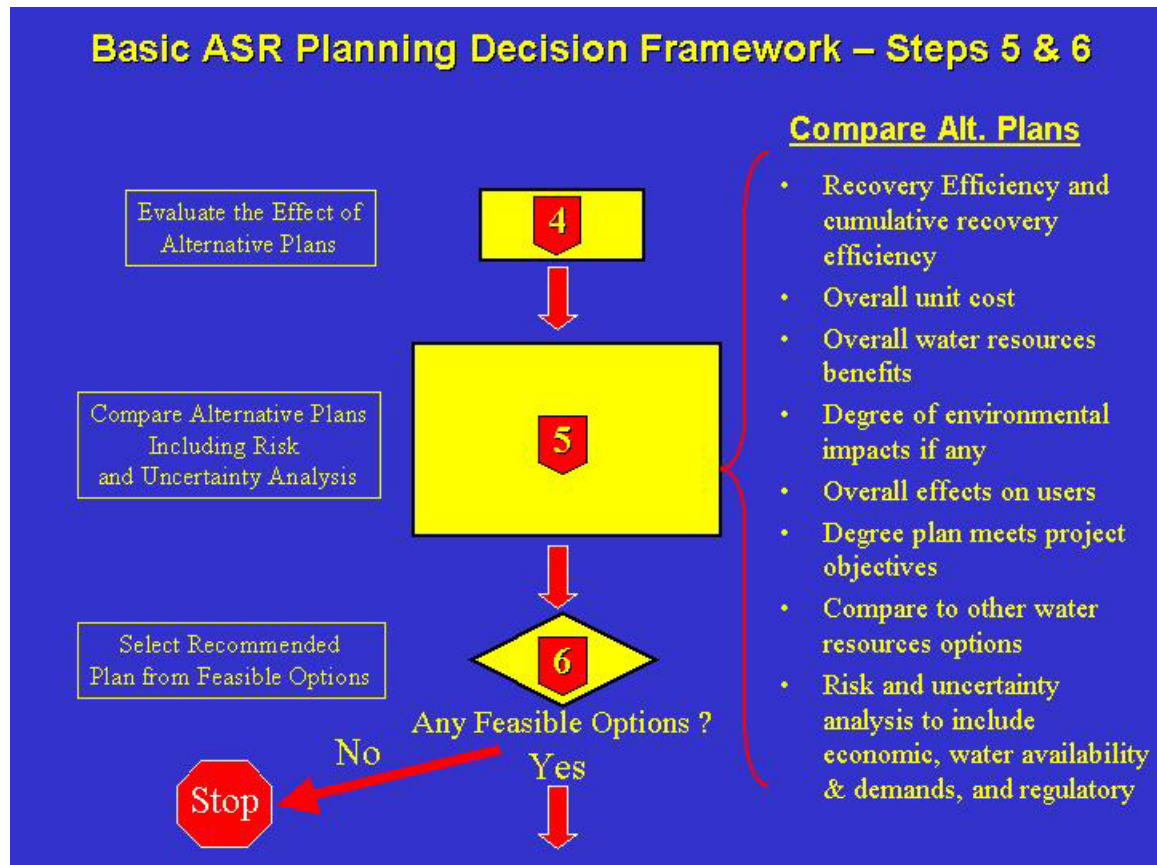


Figure 3-27. Proposed ASR Planning Decision Framework Steps # 5 and 6 with key details of evaluations required.

robust with respect to future changes in economic conditions, water availability and demands, and regulatory climate. The risk and uncertainty analysis may reveal alternatives more or less sensitive to future changes. Those alternatives that are less sensitive to future changes may be very attractive. A recommended alternative plan is then selected in Step 6 from all of the alternatives evaluated in Step 5. If no alternatives are deemed to be feasible to meet the intended project objectives, the planning decision framework should stop here and not proceed to Step 7. If, on the other hand, a viable alternative is recommended, the ASR planner must progress to Step 8 where a pilot

project plan is formulated. The pilot project plan should develop a reasonable test ASR project to be constructed and operated at the proposed project site. The pilot plan should include an ASR well, monitoring wells, a water treatment system (if necessary), and other miscellaneous site infrastructure as well as a sound testing and monitoring plan.

Obviously, regulatory coordination is also required at this step to secure the necessary permits for the pilot project. Once the plan is complete, the pilot project should be constructed and tested for some duration. One to two years is a customary pilot testing period.

Figure 3-28 provides further details on the tasks required for Steps 7 to 10 in the planning framework process. Step 9 of the process is where the pilot project results are evaluated. The ASR planner now has the opportunity to compare initial predictions to the actual observed data. Aquifer pressures, water quality changes, RE percentage, and CRE percentage must all be recorded and evaluated thoroughly. Also, unforeseen issues may arise during the pilot project. These may greatly affect Step 10 where a full-scale project is recommended or not.

Step 10 is the second major decision point in the planning framework. At this juncture, three outcomes are possible. First, the ASR planner may recommend a full-scale project to be designed based upon the recommended alternative. Second, the ASR planner may opt to recommend full-scale construction of another alternative such as a smaller overall project or perhaps another pilot project at a different location. Lastly, the pilot could have indicated that the full-scale project would not provide the benefits expected to meet the desired project objectives. The last option would mean that the

project is abandoned entirely. Even if the project is abandoned, it may be feasible in the future should conditions change or if new technology makes the project more feasible.

The basic ASR twelve step planning decision framework has been discussed for this report to provide the reader with an overview of the procedures. The actual

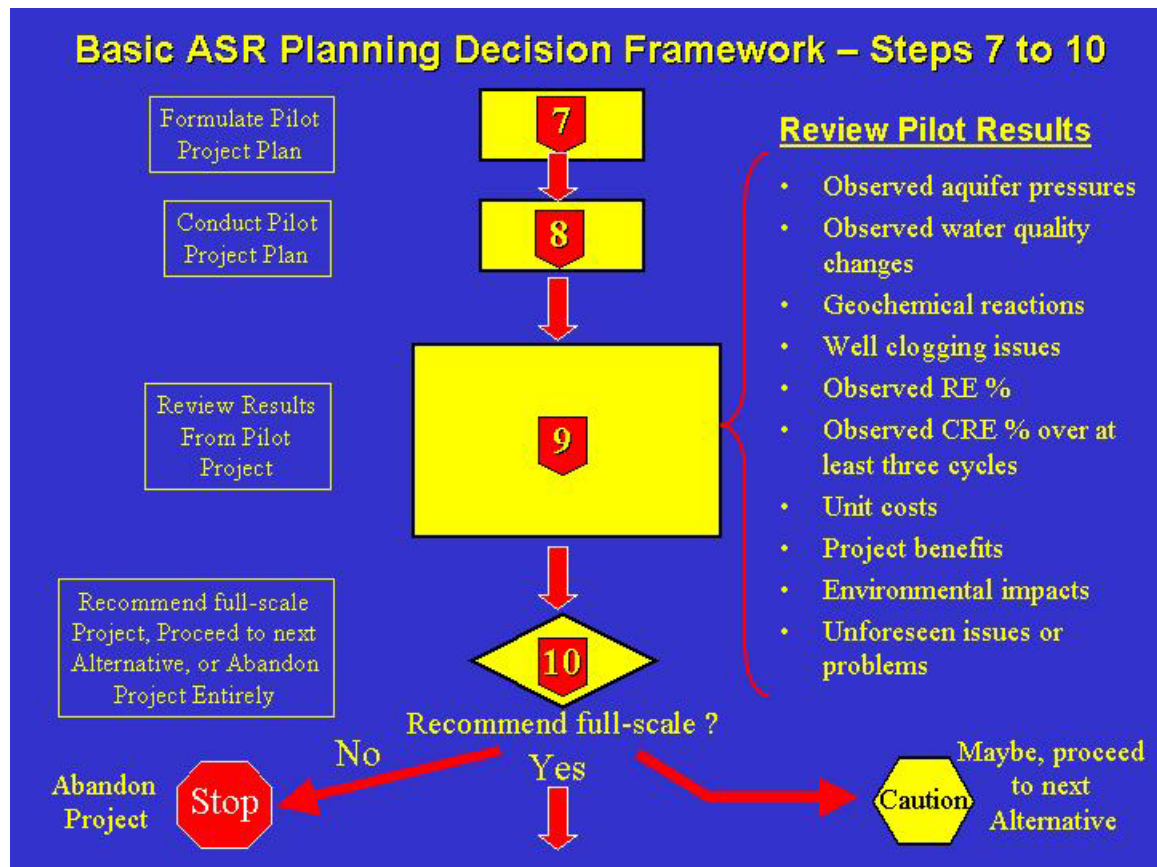


Figure 3-28. Proposed ASR Planning Decision Framework Steps # 7 to 10 with key details of evaluations required.

use of the procedure is better illustrated through its application and testing. Therefore, the proposed decision framework was applied and tested at two ASR projects from the United States. One of the sites is located in Florida while the second site is located in Texas. The two sites have not been discussed nor utilized in the development of the draft planning decision framework. The application and testing of the framework at these two

sites is discussed below. However, first, the reader is provided with a brief introduction to each of the validation sites.

Testing of Planning Framework at Two ASR Sites

Now that the basic ASR brackish water ASR planning decision framework has been developed along with new brackish aquifer performance metrics, validation of the framework must be completed. As part of this research effort, the finalized planning decision framework was tested at two existing ASR sites that were not utilized in the development of the framework. One successful ASR site and one unsuccessful site were selected for the validation effort. The successful ASR site selected for validation testing was the Edwards Aquifer ASR site (Pyne, 1995) located in Kerrville, Texas. The unsuccessful ASR site selected for validation testing was the Lake Okeechobee Taylor Creek ASR Site (CH₂M Hill, 1989a). The planning decision framework will be applied to each site in turn to determine the overall feasibility of the project. The framework will be applied as if the sites were in the early planning stages. It is hoped the validation effort will affirm the usefulness of the framework as well as provide possible sources of improvements. The application of the framework also provides a useful demonstration of its ultimate efficacy. After the validation effort, the decision framework will be adjusted as necessary from lessons learned during the validation testing process. Once minor changes have been integrated and the framework finalized, it will be recommended for use by ASR practitioners.

Overview of Two ASR Validation Sites

Lake Okeechobee, Florida, site

The Lake Okeechobee ASR project site is located in southern Florida along the northeastern portion of Lake Okeechobee. The ASR project site was an ASR

demonstration project conducted by the South Florida Water Management District (SFWMD) in 1988. The purpose of the project was to test the feasibility of storing large volumes of phosphorus-rich storm water from Taylor Creek in a subsurface aquifer thereby preventing the water from reaching the highly eutrophic Lake Okeechobee (CH₂M Hill, 1989a). The ASR demonstration project was constructed along the L-63N Canal which was a large source of phosphorus to Lake Okeechobee. The project was conceived as a single test well system that also included an onsite groundwater monitoring well (CH₂M Hill, 1989a).

Subsurface explorations revealed that the proposed ASR storage zone in the Upper Floridan Aquifer System was highly transmissive and consisted of portions of the Ocala Limestone and the Avon Park Formation (Bush and Johnson, 1988). The Ocala Limestone was encountered at 710 to 846 feet below land surface and was characterized by moderately porous fossiliferous limestone. The Avon Park Limestone was encountered at 846 to 1,400 feet below land surface and was characterized by moderately porous limestone and highly porous dolomite (CH₂M Hill, 1989b). Geophysical logging was completed including caliper, gamma ray, temperature, and resistivity. The caliper logs revealed large flow zones and voids within the Avon Park Limestone. Straddle packer testing also revealed that approximately 60% of the well flow was derived from the first major set of voids. Based upon the various test data and water quality sampling data, the ASR well was completed with an open-hole interval from 1268 to 1700 feet below land surface. The ASR storage zone was therefore located within the Avon Park Limestone. Aquifer performance tests revealed that the storage zone represented a confined leaky aquifer with a transmissivity of 572,192 ft²/day with a storage coefficient

of 1.25×10^{-3} (CH₂M Hill, 1989b). Vertical leakance (or the vertical conductivity divided by the unit thickness) was estimated to range from 0.01 to 0.001 per day.

Although abundant, source water from Taylor Creek was determined to be highly variable, with TDS ranging from 268 to 996 mg/l, chlorides ranging from 37 to 185 mg/l, TSS ranging from 8 to 19 mg/l, and total coliforms ranging from non-detectable to 7,500 per 100 ml. The ASR storage zone also had variable ambient groundwater quality since the open-hole interval is so large. Due to the large vertical interval open to the ASR well, the ambient chloride concentrations ranged from 2,000 to 3,000 mg/l and the TDS ranged from 4,000 to 6,900 mg/l (CH₂M Hill, 1989b). Due to the presence of pathogens in the source water, water treatment was included prior to ASR recharge. The water treatment system included a small detention pond for aeration of recovered water, injection pumps, and in-line chlorination. Jar testing was utilized to determine the optimal residual chlorine dose to inactivate the pathogens while minimizing the formation of disinfection by-products.

The Lake Okeechobee Taylor Creek site has generally been considered an unsuccessful ASR project (Reese, 2002; Missimer et al. 2002) due to its low initial RE percentage.

Kerrville, Texas, site

The Kerrville ASR project site is located in eastern Texas near a small reservoir and water plant situated on the Guadalupe River (Pyne, 1995). The water plant and reservoir provide supplemental water supply to the region that had been mostly dependent upon groundwater withdrawals from the Edwards Aquifer. Over pumping of the aquifer had caused a decline of groundwater levels of approximately 330 feet (Pyne, 1995). Both the

surface water and groundwater supplies are tapped to meet peak summer demands that are typically 1.53 times the average demand.

In order to provide additional peak water demand capacity, an ASR project was constructed at the water plant in 1993. The ASR storage zone is within the Lower Trinity aquifer that is characterized as a confined sandstone and conglomerate aquifer (Pyne, 1995). Aquifer performance tests provided an estimated transmissivity of 936 ft²/day and a storage coefficient of 7×10^{-4} . Leakance and porosity were estimated from rock core samples with values of 5×10^{-7} per day and 23%, respectively (Pyne, 1995). The static water level of the aquifer is approximately 130 feet below land surface while the natural pre-existing groundwater gradient was estimated to be 0.0011.

Source water quality and groundwater quality at the project site were very similar in composition (Pyne, 1995). Data presented by EPA (1999b) supports this conclusion since recharge water and ambient groundwater sampled from the Edwards Aquifer System are indeed very similar in chemical composition. The only notable exception is magnesium that has a concentration eight times higher in the ambient groundwater than the recharge water. TSS and TDS of the recharge water are very low also. Core samples collected from within the aquifer storage zone did not reveal any potential geochemical problems.

Site infrastructure includes the ASR well, submersible pump, injection tubes, and various monitoring equipment. Recharge rates are designed to range from 200 to 1,000 gallons per minute. Recovery rates that were expected at the site were in the same range as the recharge rate.

The Kerrville ASR site has been pointed to as a glowing example of a successful ASR project and is still in operation today. The project has been so successful, other ASR projects are being constructed or planned around Texas.

Testing of Planning Framework at the Two Validation Sites

Framework validation at Lake Okeechobee, Florida, site

Following the planning steps outlined in Figure 3-23:

Step 1. The Lake Okeechobee project was intended to divert nutrient-rich water from Lake Okeechobee to a subsurface storage zone in the Upper Floridan Aquifer System. The proposed project was intended to divert large volumes of water into subsurface storage so large recharge and recovery rates up to 10 MGD were envisioned.

Step 2. An inventory of available surface water within the Taylor Creek Basin indicates that water is available for recharge seasonally during the Florida rainy season. Since the primary purpose of this project is for environmental needs, the water demands for the recovered water are less important, however, it is envisioned that stored water would be recovered to the Taylor Creek Basin during the dry season or during periodic drought conditions. The source water quality is variable as presented previously with mean TDS of nearly 400 mg/l, mean chloride of 85 mg/l, mean TSS of 15 mg/l, and high levels of pathogens including coliforms. Although, blue-green algae are not specifically mentioned at this site, based upon studies discussed earlier in this report, it should be assumed that they will be present occasionally during the rainy season. The ambient groundwater within the proposed ASR storage zone is quite brackish with mean TDS of 5,400 mg/l and a mean chloride concentration of 2,500 mg/l.

The proposed ASR storage zone is within the Avon Park Formation of the Floridan Aquifer System. As discussed in the site overview, the aquifer transmissivity is

extremely high with multiple karst conduit flow zones. The transmissivity value of approximately 572,000 ft²/day is the highest value recorded out of 50 project sites reviewed for this report. The storage coefficient of 1.25×10^{-3} is slightly higher than the average value from the 50 case study projects. No dispersivity information exists for this project, therefore, a planning guideline value of 65 feet should be utilized as recommended previously. Also, given the proposed large volumes of water to be stored, scale effects should also be considered for the dispersivity estimate. Geochemical issues within the aquifer are possible given the high TSS values of the recharge water. Luckily, of the sites in Florida with existing geochemical issues, most have wells completed in the Suwannee Formation of the Upper Floridan Aquifer System or the Lower Hawthorn Aquifer within the Hawthorn Group (typically the uppermost part of the Floridan). Therefore, limited problems are expected but further explorations should be considered if a pilot project is recommended as part of this review.

The regulatory constraints are typical of projects in Florida involving in-stream projects. The recharge water must meet all requirements under Florida law regulating drinking water. The recovered water must meet all requirements for a Class III waterway. Many of these requirements are listed in Tables 3-7 to 3-10 above. Since the recharge water has pathogens in it, disinfection is required. The original disinfection plan was formulated in 1989 and included chlorination as the primary means of treatment. Based on the TSS levels and the assumed similarly high TOC levels, disinfection by-product formation in-situ would be likely. In addition, for in-stream uses, the proposed guideline value discharge limit for chloroform is 1.8 ug/l, which is extremely low given the probable level of disinfection by-products that could form in-situ. In order to

minimize potential long-term well clogging and generation of disinfection by-products, pre-treatment using filtration is recommended to reduce the TSS values to less than 10 mg/l, as recommended in this report.

Neither environmental impacts or impacts to existing users are anticipated to be problems given the location of the project within a mostly agricultural basin where surface water is used for water supply. In addition, during the wet season, most of the water is sent directly into Lake Okeechobee where it leads to further eutrophication. The well users that utilize the Floridan Aquifer System are generally at least one mile distant from the proposed project site. Therefore, set backs will not be required for this project. Geotechnical constraints should not be limiting given the extremely high aquifer transmissivity that should constrain pressure changes during recharge or recovery. Even at large recharge rates, the pressure changes within the aquifer will be minimal.

Other site selection considerations that led to the recommended location were availability of power, availability of client-owned property, and the lack of sensitive environmental habitat in the vicinity. Besides the water treatment system and the ASR well itself, additional site infrastructure should include piping, control shed, lighting, a monitoring well, and site security fencing. Due to the large recharge rate envisioned, a vertical turbine pump is recommended for the well.

Step 3. For this project, three different alternatives will be evaluated. The first alternative will store 600 million gallons of freshwater within the aquifer storage zone to be recovered in the dry season or droughts. The second alternative will store 900 million gallons of freshwater to be recovered in the dry season or droughts. The third alternative will store 600 million gallons of freshwater to be recovered for irrigation use within the

Taylor Creek Basin during the dry season. It is reasoned that alternatives one and three will utilize one 5 MGD ASR well for four months to store the required volume, while one 10 MGD ASR well will be utilized for alternative two. Assuming a reasonable RE percentage, each project is anticipated to cost \$0.75 to \$1.50 per thousand gallons recovered with alternative 2 costing towards the low end of the range. Benefits are anticipated to consist of environmental water quality improvements and water supply benefits during the dry season. Also, alternative 3 will provide irrigation benefits to the local agricultural users.

Step 4. Given the enormous aquifer transmissivity, the predicted increase in head during recharge should only be a few feet at the ASR well. Figure 3-29 provides a graphical representation of the likely head increase versus distance from the ASR well based upon the Theis analytical model presented previously. For alternative two, a head increase of twice the values shown in the figure is predicted. Even at close to five feet of increase at the ASR well, the overall aquifer pressure changes should be very small. In addition, if the confined storage zone is slightly leaky as in other south Florida projects, the overall head change would be even lower.

During recharge, water quality changes within the aquifer due to mixing and hydrodynamic dispersion are expected. The recovered water will be a blend of the recharge water and the ambient groundwater. Since the storage zone is quite brackish, it is expected that the performance of this ASR project will be less than optimum. In addition, the recovery of sulfate and sodium will need to be monitored in addition to TDS and chloride. Sodium can be toxic to many agricultural crops at concentrations over 250

mg/l. Furthermore, the recovered water will contain low to moderate levels of TTHMs and some emerging contaminants. The concentrations of HAAs are expected to be low to

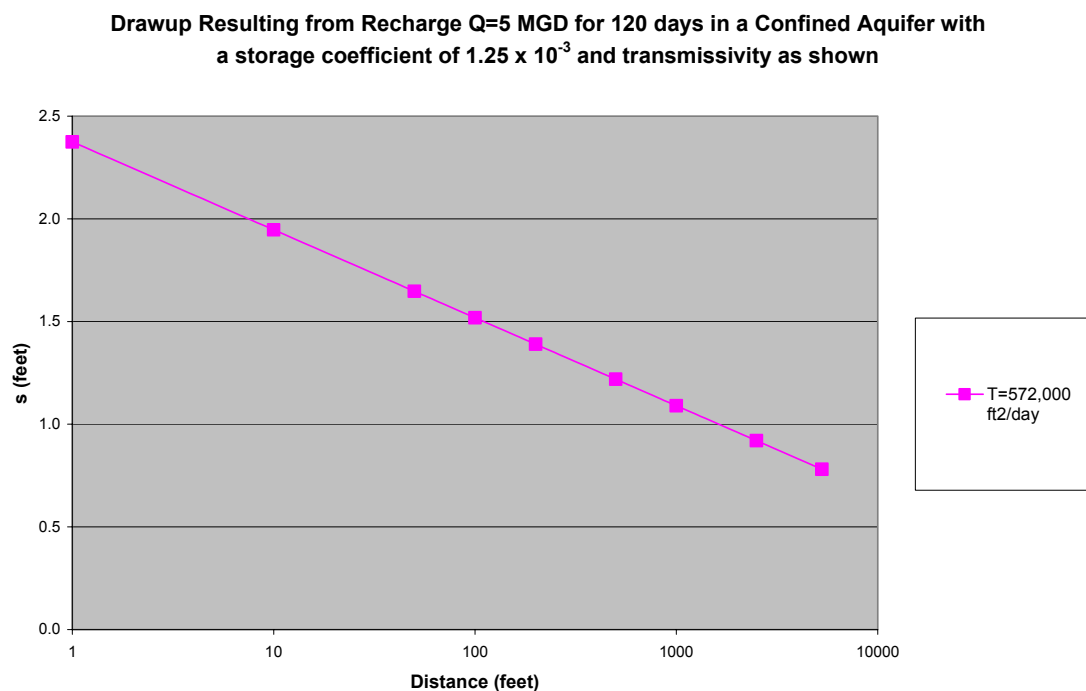


Figure 3-29. Predicted head increase from recharge rate of 5 MGD at the Lake Okeechobee Taylor Creek ASR site.

below detection limit due to expected aerobic biodegradation. The concentration of various emerging contaminants is expected to be below the levels found within the recharge surface water body within the Taylor Creek Basin due to mixing and hydrodynamic dispersion in the aquifer. The same should be true for nutrients so that a net water quality improvement for these categories is expected for some time. No acute or chronic toxicity effects are expected related to ecological receptors, however, toxicity testing is recommended should a pilot project be completed. Temperature changes within the surface water are expected and will have to be mitigated using a mixing zone. Temperature changes in the Taylor Creek Basin due to recovery of warmer stored ASR

water could be problematic during extreme drought events. More detailed engineering evaluations should be considered during any recommended pilot project.

Step 5. In comparing the three alternatives developed for Taylor Creek, it is important to estimate the ASR RE percentage and long-term CRE percentage. In order to develop the estimates, the recovery index and inverse relative dispersivity dimensionless parameters will be calculated for each alternative. For the three alternatives, all information necessary to calculate the dimensionless parameters is available except the dispersivity. As discussed in this report, two approaches are recommended. First, a suggested planning guideline value of 65 feet for the dispersivity can be used since the geologic character is aptly described as karst limestone and the project scale is assumed to be of secondary importance. Second, the dispersivity can be estimated based upon the geologic character and the scale of the project. Using equation 6 of this report and replacing “L” with r_m , a reasonable second estimate of dispersivity can be approximated at 78 feet for alternatives one and three while the power law model estimates a dispersivity value of 101 feet for alternative two. Both of the estimated dispersivity values can then be used as input into the dimensionless parameters in order to develop estimates of RE percentage. Then both planning tools can be utilized to provide RE estimates. Both the linear regression model # 1 and the performance envelopes are available to assist in this task.

Table 3-33 provides a summary of the recharge volumes for each alternative, the dimensionless parameters, and the estimated initial cycle one RE percentage. The first three estimates utilize the dispersivity values calculated from the power law equation 6.

The second three estimates utilize the dispersivity guideline value of 65 feet for a karst limestone aquifer.

As the table clearly demonstrates, the initial cycle one RE percentage is predicted to be zero using both proposed planning approaches. Using the graphical method as presented in figure 3-30, the RE estimate is five percent. According to the performance envelope, even if the dispersivity estimate were lower, the maximum initial

Table 3-33. Taylor Creek ASR alternatives 1, 2, and 3 with predicted RE percentage.

Alt #	Recharge Volume (ft ³)	Aquifer thickness b (feet)	Porosity %	r _m (feet)	LN (R _α)	LN (RI)	RE estimate %
1	80,213,904	432	25	486	1.95	2.72	0
2	160,427,808	432	25	688	2.06	2.72	0
3	80,213,904	432	25	486	1.95	2.72	0
1	80,213,904	432	25	486	1.95	2.72	0
2	160,427,808	432	25	688	2.06	2.72	0
3	80,213,904	432	25	486	1.95	2.72	0

cycle RE percentage would be 16%. Taking the mean of the three RE estimates, the estimated RE value would be 1.67 %. Now this value can be utilized in one of the CRE charts to estimate long-term performance. Figure 3-31 presents such an estimate for Taylor Creek. Although the original chart was developed for an aquifer with chloride of 2,000 mg/l and recharge water quality (RWQ) of 5 mg/l, the same chart can be used to provide a crude long-term performance estimate if it is understood that the real performance would probably be a bit lower since the real AWQ is actually 2,500 mg/l chloride and the RWQ is closer to 85 mg/l. In any event, it is plain to see from the figure that after six cycles, the CRE would still be less than 30% and the recovery curve would be approaching an asymptote similar to what has been observed at other ASR projects. Furthermore, in the project assumptions, water storage would occur in the wet season

only for 120 days, followed by recovery in the dry season or during droughts. If the storage period is longer than 90 days, additional buoyancy stratification effects may further reduce the CRE percentage.

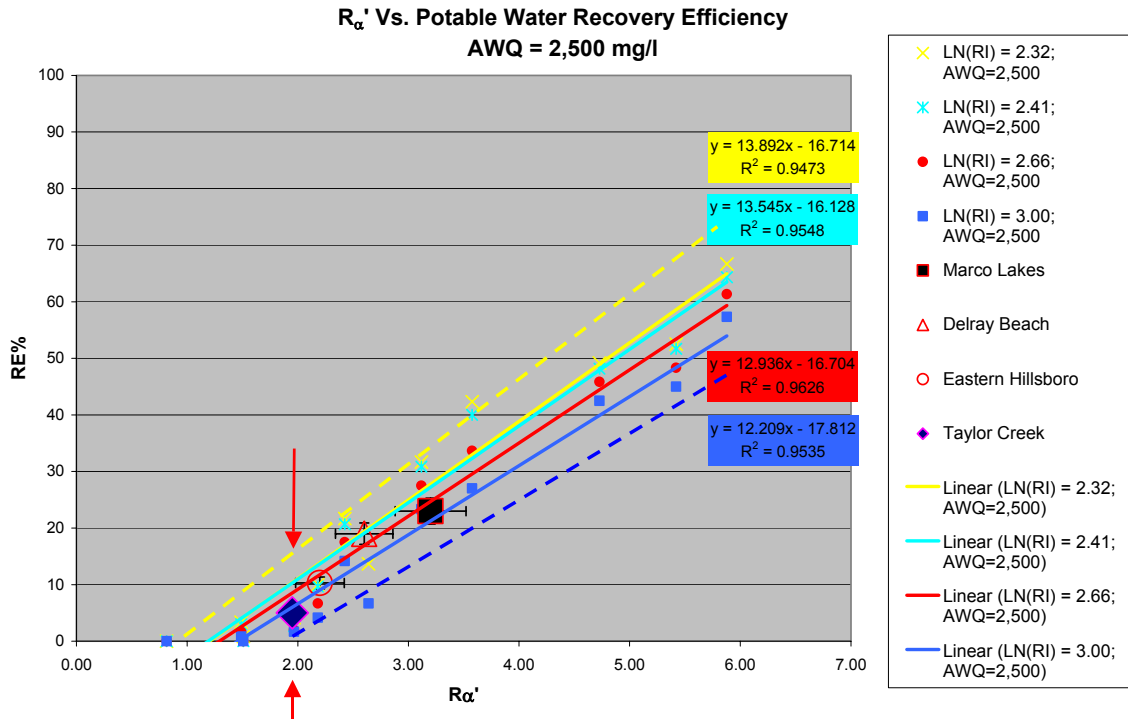


Figure 3-30. RE estimate for the Taylor Creek ASR site using graphical method.

Obviously, the poor recovery affects both the unit cost of the project and the benefits. As has been pointed out previously, the unit cost can increase rapidly as the RE percentage or CRE percentage decreases. Similarly, the benefits also decrease substantially. Due to the low CRE values, the estimated unit cost would be \$2.70 to \$3.00 per thousand gallons water recovered. At this unit cost, other water resources options including reservoirs or wastewater reuse or just water treatment may be more economical. Since the main benefit of alternatives one and three is water quality improvements to the Lake Okeechobee, these options may still provide limited positive

contributions. Alternative 2 probably would not be feasible at all since the water supply provided would be insufficient as a dependable irrigation supply.

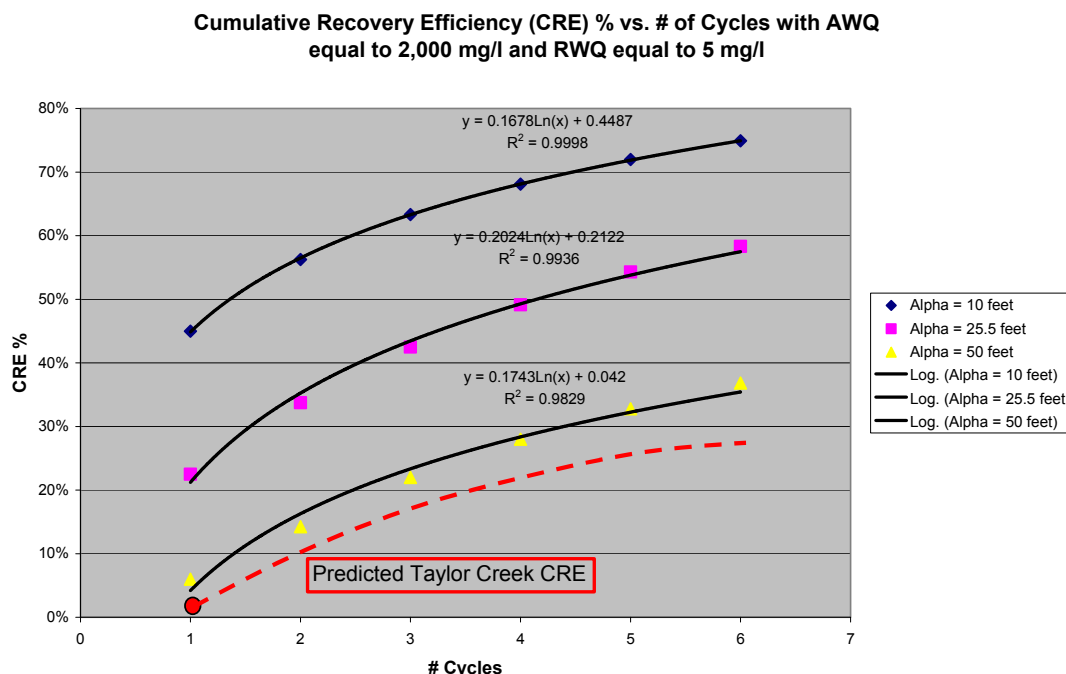


Figure 3-31. CRE estimate for the Taylor Creek ASR site using graphical method.

To complete the remaining alternative comparisons, the water quality of the recovered water for each alternative should be weighed for environmental impacts. The recovered water quality for irrigation use would not be problematic, however, the recovered water for in-stream use may add significant amounts of chloride, sulfate, sodium, and disinfection by-products to the surface water. The recovery of chloroform (and other DBPs) would likely have concentrations greater than the 1.8 ug/l ecological guideline value. If a mixing zone could be employed, this issue might be mitigated, however, the in-stream objective is to return the recovered water during the dry season or during droughts; thereby the mixing zone may not be as helpful.

As far as risk and uncertainties go, the main uncertainty is the dispersivity that will occur, however, the analysis already used a reasonable range of estimates. Another

uncertainty is the risk of more stringent regulatory requirements in the future for DBPs. The chloroform guideline value is not a legal requirement in the United States, however, it may become one in the future. For economic risks, the main issue would be the possibility that the CRE percentage is even lower than 30%. This is possible and would push the unit cost to more than \$4.00 per thousand gallons water recovered for each of the three alternatives.

Based upon application of the newly proposed ASR planning decision framework, the Taylor Creek ASR project seems to be infeasible for all three alternatives evaluated. Proceeding to Step 6 of the framework, it is recommended that a different location within the Taylor Creek region be evaluated so that the project may proceed to a pilot project phase.

Framework validation at Kerrville, Texas, site

Step 1. The Kerrville ASR project is intended to provide peak demand water supplies for the area around Kerrville, Texas. The project was planned to recharge 200 to 1,000 gpm during off peak demand times.

Step 2. As was previously discussed in the site overview, surface water is available during offpeak periods in the fall and winter. The surface water quality is actually similar to the ambient groundwater quality with the exception of magnesium. Magnesium can be elevated within the Edwards Aquifer in Texas. Generally, however, the two water quality end-members are almost identical and compatible. The site is considered a non-brackish ASR site since the existing TDS level in the aquifer is less than 1,000 mg/l. A suitable aquifer is available at the site and since the aquifer has been over-allocated in the past, water levels have been lowered by over 300 feet from historic levels. This provides an excellent opportunity to refurbish the aquifer storage potential through the use of ASR

recharge. Currently, the aquifer water levels are 130 feet below land surface. The confined sandstone aquifer storage zone has an aquifer transmissivity of 936 ft²/day and should be ideal for an ASR project of small to moderate magnitude.

Geochemical issues should not be a problem based upon the source water and ambient groundwater similarities and testing of aquifer materials that indicated no potential problems within the aquifer matrix itself. Regulatory constraints should not be a problem at the site since the water quality within the aquifer is similar to that of the source water. Certainly pathogen reduction will be required prior to recharge into the aquifer since the literature indicates that most surface waters contain various concentrations of pathogens. Geotechnical constraints may be important at the Kerrville site since the transmissivity is low as compared to the mean value determined through a review of 50 ASR sites from around the world. Environmental constraints should be minimal since the recovered water will be directed to potable uses in the study area.

The site infrastructure required includes an ASR well, monitoring wells, water treatment plant, piping, and pumps. In the case of the Kerrville project, an existing water treatment plant is the site of the project and was selected as such due to unused water treatment capacity that is available during offpeak water supply demand periods.

Therefore, the cost of this project is expected to be low.

Step 3. In the plan formulation step, two alternatives will be considered for the project. Both the alternatives are potable water supply alternatives since that serves the objectives of the project. The first alternative will endeavor to store 25,000,000 gallons of freshwater with a recharge rate of 0.25 MGD for 100 days during the fall and winter. The second alternative will endeavor to store 50,000,000 gallons of freshwater with a

recharge rate of 0.50 MGD for 100 days during the fall and winter. The existing water treatment plant includes conventional coagulation, filtration, and chemical disinfection. Therefore, the TSS values are generally less than 1.5 mg/l and the TOC levels are also low. TTHMs and HAAs are low to moderate and additional formation is not expected in the aquifer. The costs of the project are expected to range from \$0.25 to \$0.50 per thousand gallons recovered. This cost would be extremely economical.

Step 4. The potential effects of the two alternatives should not be great since both alternatives are just utilizing the full water treatment plant capacity for a longer duration during the year. The amount of water being allocated to the project will not significantly change the existing water budgets within the river basin. Furthermore, the recovered water will contain low to moderate levels of TTHMs and some emerging contaminants. The concentrations of HAAs are expected to be low to below detection limit due to aerobic biodegradation expected. The concentration of various emerging contaminants is expected to be below the levels found within the recharge surface water body within the river basin due to mixing and hydrodynamic dispersion in the aquifer. Geochemical reactions are not expected to result in any impacts or effects. Of all of the potential effects, pressure changes in the aquifer are expected to be the main source of effects. Figure 3-32 provides an estimate of the pressure changes expected within the Edwards Aquifer from the proposed Alternative 1. As can be clearly seen, the maximum head change at the ASR well is approximately 55 feet. For alternative 2, the maximum head would double to 110 feet. Either alternative should be feasible given that the existing water level is 130 feet below land surface.

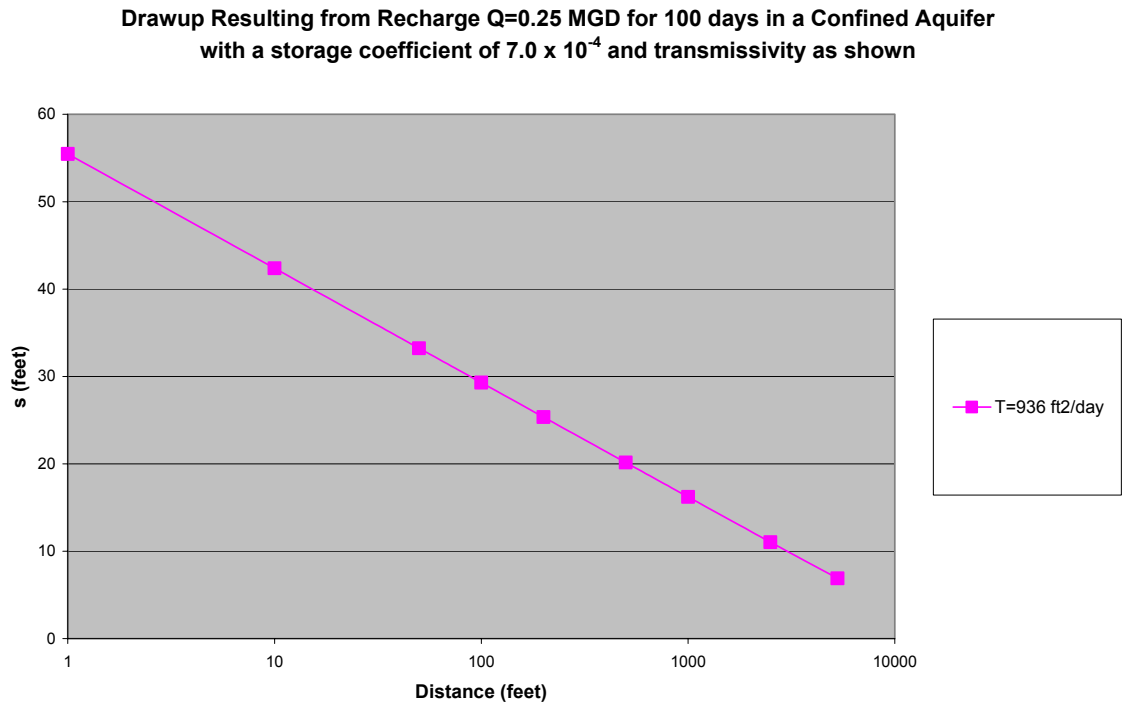


Figure 3-32. Expected head increase from Kerrville alternative 1.

Step 5. Either alternative appears quite feasible but since the estimated unit cost of alternative 2 is expected to be lower, this is the alternative that should be recommended. Alternative 2 should be feasible under any number of expected future conditions since the alternative is very cost effective.

Step 6. Alternative 2 is recommended for pilot testing and further engineering study.

Amendments to Proposed Planning Decision Framework

The testing of the framework revealed that it is sound in formulation and is flexible. For example, the Kerrville site is a notable ASR project due to its successful implementation. The framework also concluded that the project was most likely feasible and should be tested further during pilot testing. The Taylor Creek ASR project is a brackish water project. The framework is probably more useful for brackish water sites

since new metrics and performance measures have been developed as part of this research effort for those sites. Testing the framework at the Taylor Creek site determined that the project would not be very feasible and that another project location might be more suitable. One weakness in the framework is potential ASR sites where the ambient groundwater is slightly brackish with TDS levels greater than 1,000 mg/l but chloride levels are lower than 250 mg/l. The newly developed brackish water ASR performance metrics are designed for sites where the ambient chloride concentration is greater than 250 mg/l. Therefore, brackish water sites that do not fit this description would have to be treated similar to the Kerrville site. Based upon the testing of the framework, no changes are recommended and it should be utilized for evaluation of new ASR projects. As part of a final demonstration of the framework, it was utilized to evaluate the proposed Everglades Restoration project ASR pilot project sites.

Application of New Planning Decision Framework at CERP ASR Pilot Sites

Now that the new ASR planning decision framework has been formulated, tested, and then amended, it is ready to be utilized at new prospective ASR projects. The ASR Pilot Projects, proposed as part of the Everglades Restoration (USACE and SFWMD, 1999), represent a worthy candidate for final testing. This portion of the report first discusses the Everglades restoration and the ASR Pilot Projects. Finally, the new ASR planning decision framework, developed as part of this research project, is applied at each prospective ASR pilot site in order to predict performance and make further planning recommendations in regard to the pilots.

The Everglades Ecosystem, located in Southern Florida, is composed of an amalgamation of wetlands, tidal marshes, cypress domes, estuaries and coral reefs. The Everglades is a unique ecosystem, like no other in the world. It is a broad, flat expanse of

wetlands inhabited by a myriad of plants and animals. Dubbed the “River of Grass” by Marjorie Stoneman Douglas (Douglas, 1947), the Everglades is in trouble. The distribution of water, its timing, its quality and its quantity, have all been radically changed over the last 100 years (Davis, 1994).

The Central and Southern Florida Project Comprehensive Review Study (USACE, 1999), developed jointly by the South Florida Water Management District (SFWMD) and the U.S. Army Corps of Engineers (USACE), presents a framework for Everglades restoration. Now known as the Comprehensive Everglades Restoration Plan (CERP), this plan contains 68 components, including structural and operational changes to the Central and Southern Florida (C&SF) Project. The overarching purpose of CERP is to restore the Everglades by improving the quantity, quality, timing and delivery of water for the natural ecosystems of south Florida. A key component in the overall restoration strategy is the provision of more dynamic storage of freshwater. Freshwater is a necessary ingredient to life in the Everglades, however, the supply of such water is limited and finite. Many researchers lament the existence of a “water crisis” in the world (Bader & Ahmad, 2000; Rodda, 2001). In addition, competing demands for water supply (irrigation, drinking water, industrial cooling, in-stream environmental uses, etc.) have led to political upheaval and multiple lawsuits. In the Southeastern United States, competing water demands in various river basins threaten domestic tranquility (Feldman, 2000).

The various pressures have provided opportunities to explore new water supply technologies as part of the overall CERP. One of the technologies proposed is Aquifer Storage and Recovery (ASR) wells. The use of ASR is increasing in the United States

and abroad. The CERP relies heavily upon ASR technology to provide additional system storage. The proposed use of ASR for this project is unprecedented with 333 large-capacity wells proposed to store over 3.8 million cubic meters of water per day. Six ASR components currently form the proposed CERP ASR System, which includes a total of 333 ASR wells and related surface facilities at the general locations in attached Figure 3-33.

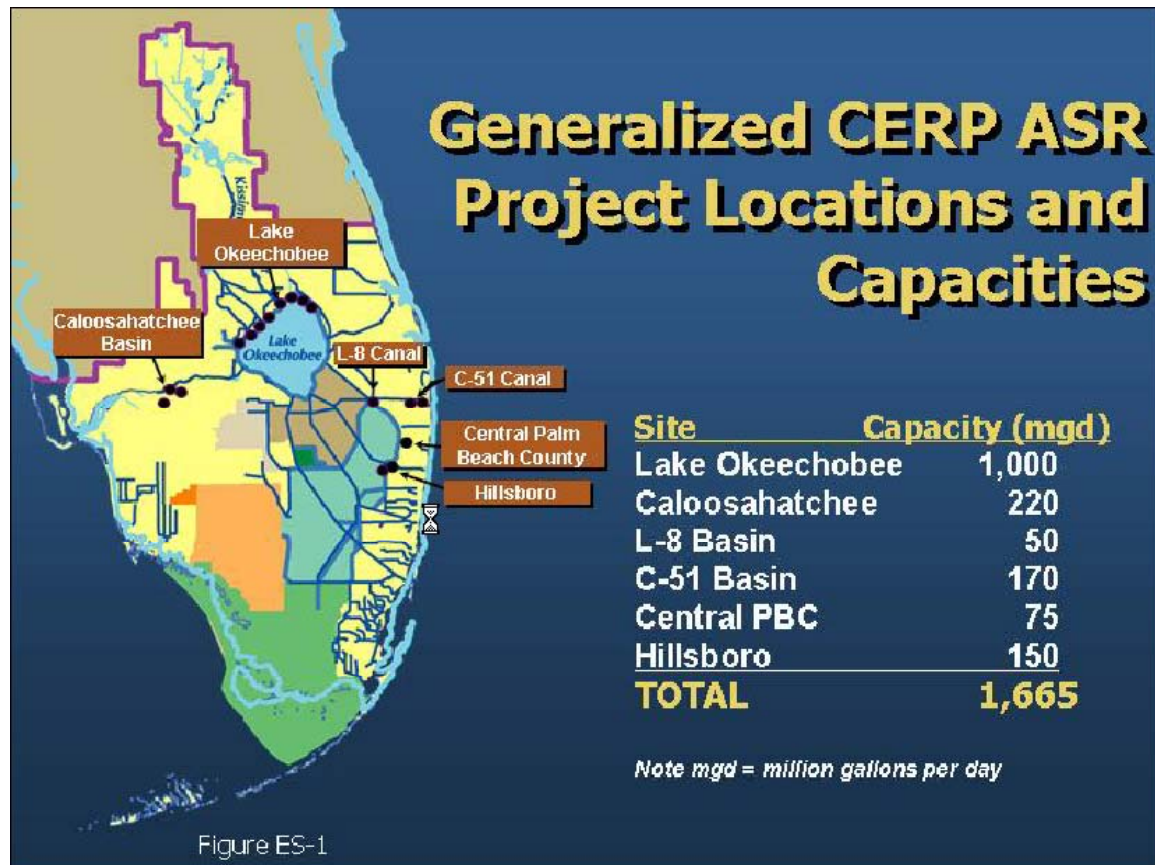


Figure 3-33. Location and capacity of proposed CERP ASR Wells in southern Florida.

All proposed ASR wells have a target capacity of 5 million gallons per day (mgd). Water treatment facilities were also included in the conceptual CERP ASR components. Total cost of the proposed CERP ASR System is approximately \$1,700,000,000. The United States Army Corps of Engineers (USACE), Jacksonville District and the South

Florida Water Management District (SFWMD) are 50/50 cost sharing partners for design studies required prior to implementation of any large-scale CERP ASR facilities.

Local-scale studies have been initiated in the form of three ASR Pilot Projects as recommended in the CERP. The location of the five separate pilot sites (part of three ASR pilot projects) is shown in Figure 3-34.

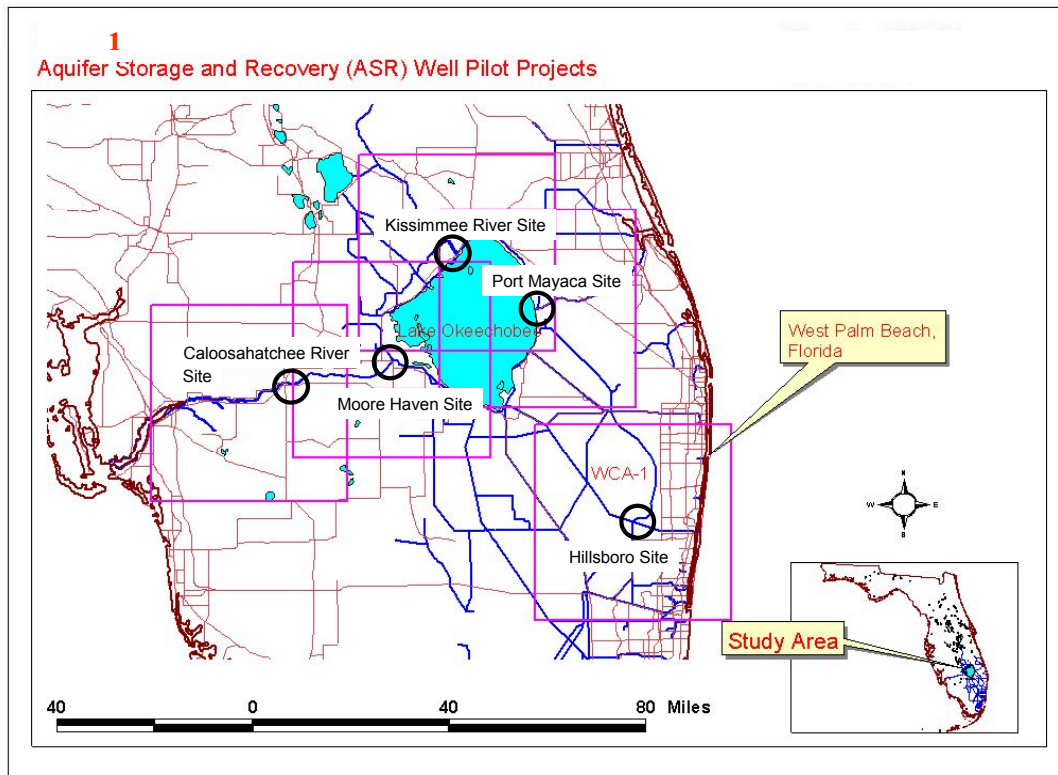


Figure 3-34. General location of the CERP ASR Pilot Projects.

The CERP Lake Okeechobee ASR Pilot Project will initially consist of up to five ASR wells, each with an estimated capacity of 5 mgd (USACE, 2003). Three of the ASR wells will be located spatially around Lake Okeechobee to demonstrate ASR performance in geographically dispersed areas. A single three-well cluster facility will be installed at one of these locations to demonstrate how a multiple-well ASR system performs. The CERP Caloosahatchee River ASR Pilot Project and CERP Hillsboro ASR

Pilot Project each will consist of one 5-mgd ASR well. Monitor wells and surface facilities will also be constructed at each of these systems. The ASR wells will recharge and recover surface water from the lake and/or its tributaries. Extensive water quality characterization and pilot treatment testing has taken place during the permitting and design phase of these projects. Once constructed, the CERP ASR pilot project systems will be cycle tested to evaluate their ability to achieve assumed water quality and volumetric levels of performance, and allow for recommendations to be made for facility expansion.

The hydrogeology and geology of each site is somewhat similar. South Florida is underlain by Cenozoic-age rocks to a depth of approximately 5,000 feet below land surface, comprised primarily of sand, limestone, clay and dolomite (Meyer, 1989). Within this province, Lake Okeechobee lies in a relatively stable structural area, represented by generally flat-lying sediments that accumulated in a quiet marginal-marine setting, similar to the modern-day Bahamas. Numerous wells have been constructed and tested to depths of up to approximately 3,500 feet bls in south Florida, providing rather extensive information regarding subsurface geology and hydrogeology of the area.

The hydrogeology in most of South Florida consists of a layer cake of aquifers and confining units. The three primary aquifers in the study area include the Surficial Aquifer System (SAS) ranging from 100 to 300 feet thick; the Intermediate Aquifer System (IAS) located within the Hawthorn Group sporadically; and the massive Floridan Aquifer System (FAS) that can be as thick as 1,500 feet. Generally, the SAS is separated from the FAS by an extensive confining unit consisting of interbedded sands, clays and carbonate units. The Intermediate Confining Unit (ICU) occurs between 150 and 850

feet below land surface in the study area and is usually synonymous with the Hawthorn Group. The FAS can generally be subdivided into several permeable zones, separated by low-permeability limestones. It is composed of limestone and dolostone units generally dipping to the east and south, and contains brackish to saline water. The permeable zones within the FAS are regionally grouped into upper and lower units, separated by a middle confining unit. These units are informally designated "Upper Floridan Aquifer", "Middle Floridan Aquifer Confining Unit", and "Lower Floridan Aquifer" (Miller, 1997). A general hydrogeologic across south Florida (prepared by the USACE and the SFWMD) is presented as Figure 3-35.

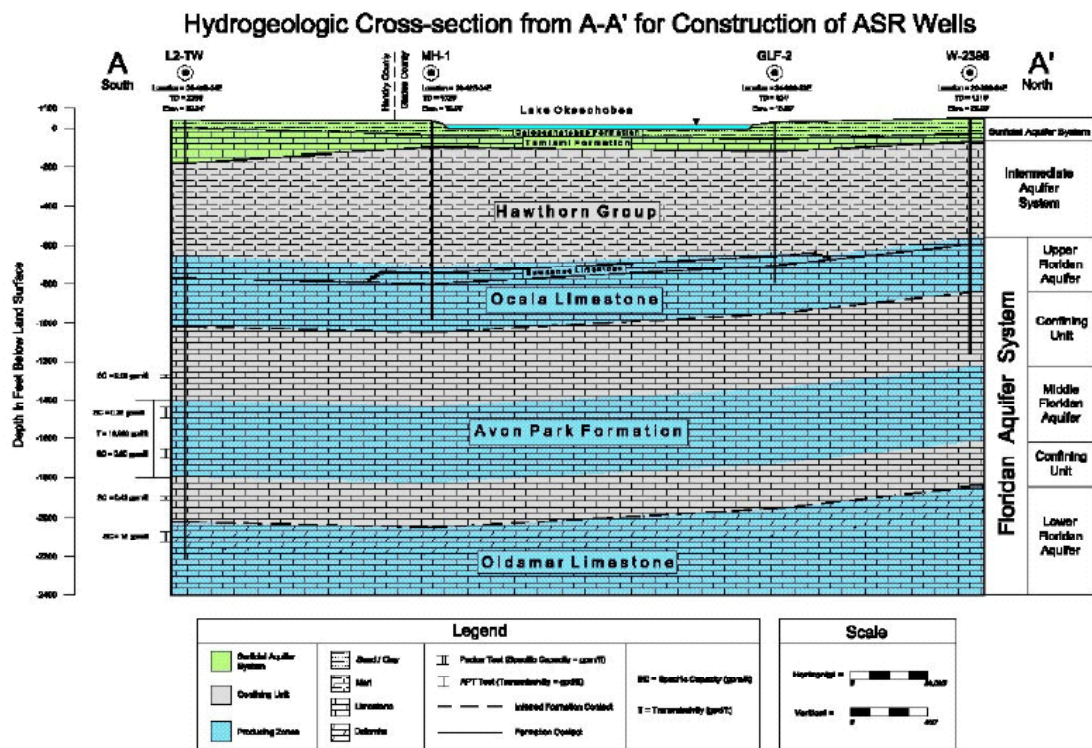


Figure 3-35. Location and general hydrogeology at the Hillsboro ASR pilot project.

In support of the ASR Pilot Projects, extensive field investigations were conducted to gather important hydrogeologic and water quality data to support project planning and design. Test Wells were drilled at the Hillsboro site (SFWMD, 2001), Port Mayaca

(USACE and SFWMD, 2003), Kissimmee River (USACE and SFWMD, 2004), Moore Haven (USACE and SFWMD, 2003), and near the Caloosahatchee River (Bob Verrastro, 2004).³ Investigations at the Caloosahatchee site are ongoing and will be completed this year. Site-specific investigations included collection of soil and rock cores; downhole geophysical logging; packer testing; aquifer testing; and water quality sampling. Table 3-34 provides a summary of key supporting information developed through the field investigations.

Table 3-34. CERP ASR Pilot Projects key supporting technical data.

Pilot Site	Transmissivity	Recharge Water mean chloride concentration (mg/l)	Ambient Groundwater mean chloride concentration (mg/l)	Geologic Character
Hillsboro Canal	648 m ² /day	150	1,900	Limestone with some small conduits
Port Mayaca	1,180 m ² /day	100	850	Limestone with some small conduits
Kissimmee River	845 m ² /day	100	400	Limestone with some small conduits
Moore Haven (deferred due to financial constraints)	<800 m ² /day, est. from Sp. Capacity Test	100	2,400	Limestone with some small conduits
Caloosahatchee River	<800 m ² /day, est from Sp Capacity Test	75	700 mg/l estimated from nearby Well Lab-TW	Sandy Limestone

It is obvious that the variation in hydrogeologic properties across these sites is important to the overall planning effort. None of the sites exhibit a karst limestone geologic character. Four of the sites could probably be characterized as limestone with minor

³ Personal communication from Mr. Bob Verrastro of the SFWMD

fractures and conduits. The Caloosahatchee River site has a very sandy limestone character. In order to assign dispersivity values to each site, both estimating approaches described in this report were attempted. Based upon the geologic character alone and assuming recharge of 5 MGD for 30 days for all project sites, the dispersivity of the first four sites would be assigned a value of 30 feet, whereas, Caloosahatchee would be assigned a value of 7.5 feet due to its sandy nature. Using the power law approach and an estimated r_m value of approximately 360 feet (based upon 200 feet thick storage zones at all sites), the dispersivity is estimated at approximately 47 feet for the first four sites, whereas Caloosahatchee would be approximately 30 feet. All pilot sites also exhibit ASR storage zones that contain brackish water. In general, for brackish water ASR sites, a complex interplay between the source water quality and the ambient groundwater quality may control the project feasibility. The transmissivity of the upper FAS calculated at each exploration site indicates that the magnitudes are at the low end of the range quoted by Bush and Johnston (1988). Bush and Johnston estimate a transmissivity range from approximately $800 \text{ m}^2/\text{day}$ to $4,800 \text{ m}^2/\text{day}$ for the upper FAS. Although not listed in the table, the storage coefficients appear to be similar to those estimated by Bush and Johnston (e.g., 1.0×10^{-3} to 1.0×10^{-4}).

Further site-specific information concerning each pilot site is detailed in the following sections of this report. The additional site-specific information in combination with a solid understanding of the geology and hydrogeology will permit application of the new planning decision framework to all five pilot project sites. For this report and decision framework application, all five potential pilot project sites will be treated as individual alternatives to be evaluated and compared against each other. The recharge

and recovery rate will be assumed to be 5 MGD for each site and the recharge and recovery durations will be 30 days each. Therefore, each cycle will assume storage of 150,000,000 gallons of freshwater. Recovery will end once the chloride concentration of the recovered water reaches 250 mg/l. The following paragraphs provide a brief overview of each pilot site along with site location maps. After the overviews, the final planning decision framework is applied from Step 1 to Step 6. For brevity sake, the pilot projects are grouped together in the framework application discussion. The actual pilot projects are scheduled to begin construction in early 2006. It is hoped that by applying the decision framework, the pilot project locations can be validated as reasonable pilot project candidates.

The Hillsboro ASR Pilot Project is located along the Hillsboro Canal in southern Palm Beach County. More specifically, it is located at the southern end of the LNWR, also known as Water Conservation Area (WCA) No. 1, where it intersects with the northeastern corner of WCA No. 2A. Site coordinates are Latitude N 26° 21' 07", Longitude W 80° 17' 22". The site is located west of Boca Raton, Florida, about 6 miles west of State Road 7 (U.S. Hwy 441) at the end of Loxahatchee Road (State Road 827). The project is on the north side of Loxahatchee Road and the Hillsboro Canal. An unpaved access road parallels the north side of the canal along the canal bank. The access road can be reached by crossing either the S-39 Structure (to the west) or a bridge over the Hillsboro Canal about 4,000 feet east of the site. Figure 3-36 provides an aerial photo of the project location in southern Palm Beach County, Florida.

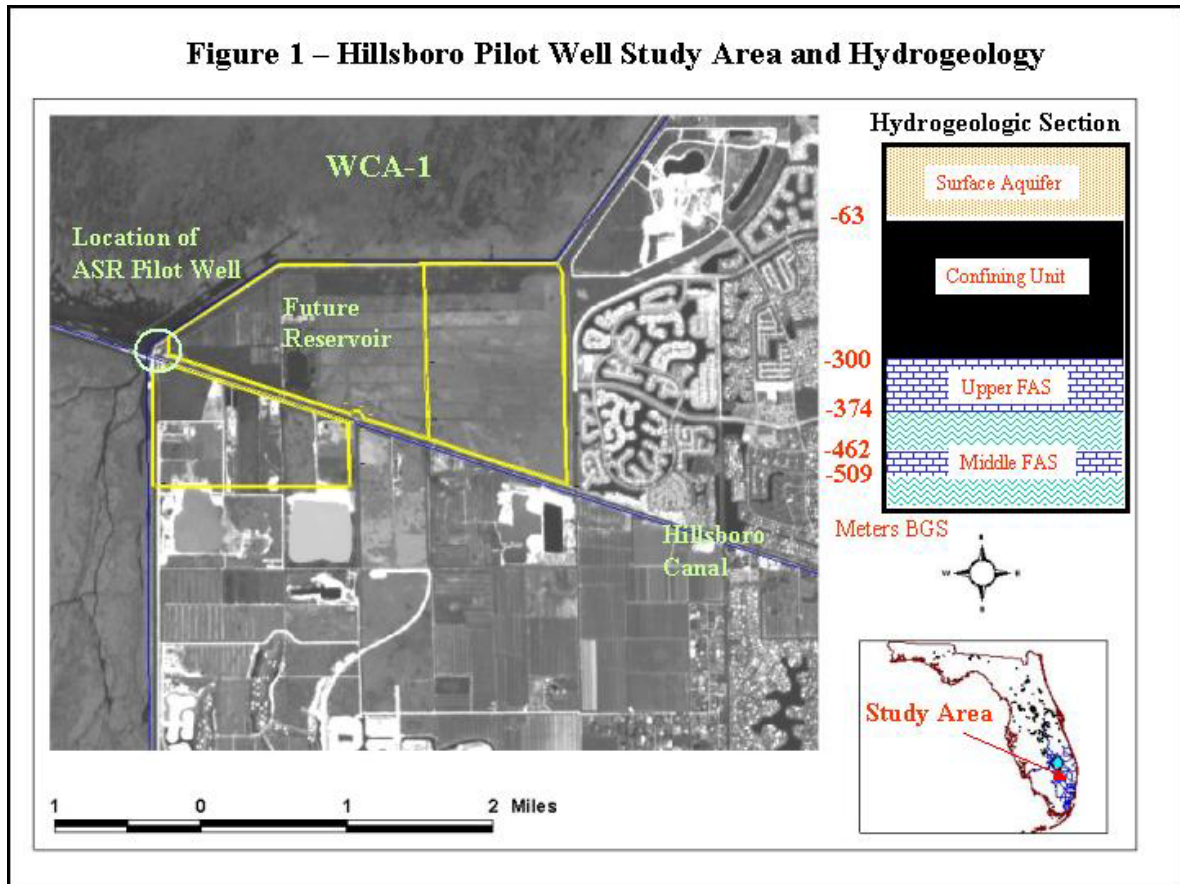


Figure 3-36. Location and general hydrogeology at the Hillsboro ASR pilot project.

The ASR pilot project site is located at the western corner of the Site 1 property owned by the SFWMD. The 1,660-acre Site 1 tract has been used for improved pasture, nursery stock and aggregate mining. Site 1 is intended to be used as an above-ground water reservoir as part of the CERP. Three of the features at the western corner of the property include a two-acre pond/lake, and a communications tower, and the S-39 control structure. The geology and hydrogeology of the site is discussed by SFWMD (2001).

The Port Mayaca site is located east of Lake Okeechobee in the northwest corner of Section 14, Township 40 South, Range 37 East, near the confluence of the L-65 Canal and St. Lucie River (C-44 Canal) in the Town of Port Mayaca, Florida. The site is located on a SFWMD-owned parcel of land adjacent to the S-153 spillway and lock,

which conveys water to and from the L-65 Canal and the St. Lucie River. Site coordinates are Latitude N 26° 59' 17", Longitude W 80° 36' 22" (see Figure 3-37).



Figure 3-37. Location of the Port Mayaca ASR pilot site (source SFWMD).

This location is approximately 2,000 feet east of the Herbert Hoover Dike, south of the service access road, and approximately 100 feet west of the intersection with the L-65 canal. Property to the north is under sugar cane cultivation. A site aerial photo is presented in Figure 3-37. The site includes a canal berm and two access roads that form swales between and on either side, separated from adjacent agricultural property by a vegetated drainage canal. The canal berm slopes steeply approximately 10 feet to the canal surface water. This slope is not stabilized and is eroding. Soils are the result of dredge material and erosion. The entire area has been impacted by use and maintenance of the dike, levee, and lock system. Accordingly, it is doubtful any historically or culturally significant artifacts or sites remain or could be recovered. The USFWS has

characterized the existing habitat as marginal to poor for wildlife utilization due to the highly disturbed nature of the area.

The Kissimmee River site is located in the northwest corner of Section 19, Township 38 South, Range 35 East, near the confluence of the Kissimmee River and Lake Okeechobee near the Town of Okeechobee, Florida. The site is located on a SFWMD-owned parcel of land adjacent to the banks of the Kissimmee River. A site map is presented in Figure 3-32. The site is approximately 8,000 feet upstream from the river's connection to Lake Okeechobee and north of Route 78. Specifically, it is located at latitude N 27° 09' 18.7", longitude W 80° 52' 29.7". The site location is depicted on an aerial photo as Figure 3-38.

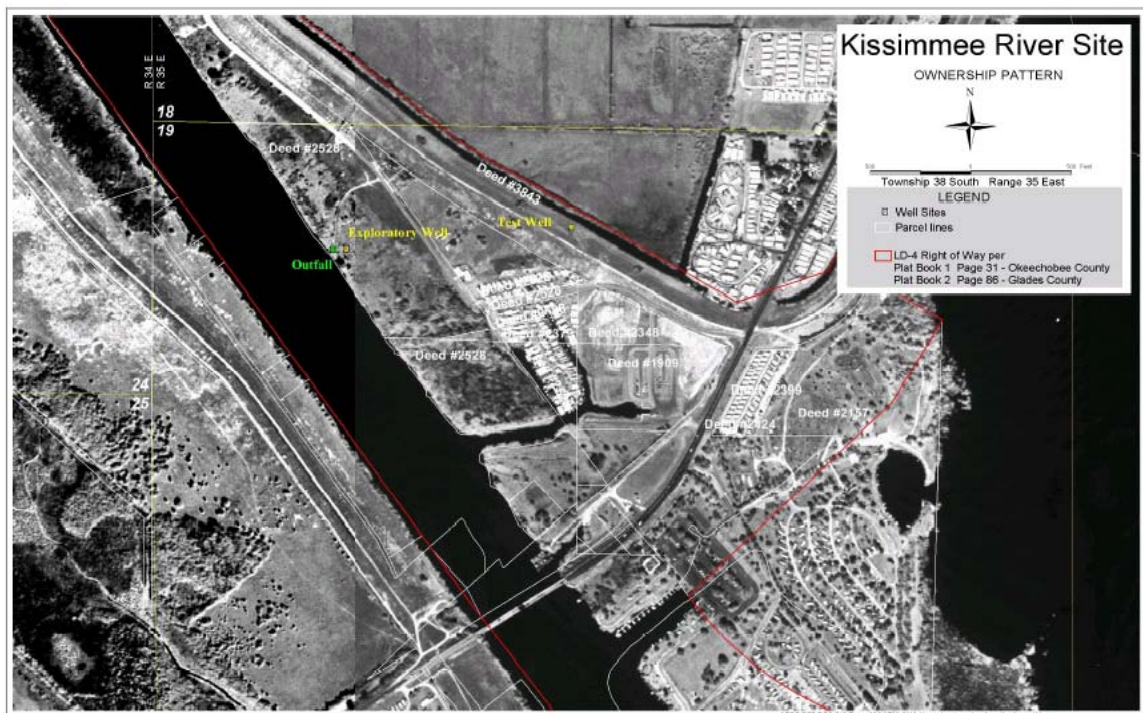


Figure 3-38. Location of the Kissimmee River ASR pilot site (Source SFWMD).

The site is in a grassy field situated on the east side of the Kissimmee River and south of an access trail. The area is bordered on all sides except the east by similar, undeveloped lots. The east boundary is a paved road with a residential home on the east

side of road. Observed soil conditions appeared as highly disturbed, sandy spoil material, presumed to be from side casting of Kissimmee dredging and channelizing projects. The site consists mainly of tall grasses and brush areas that could provide food and protection for mammals, reptiles, and bird species. The adjacent section of the Kissimmee River is an important spawning area for black crappie and other recreationally and commercially important fish species. Due to the extensive impact to the area by past activities, it is unlikely that any historical or culturally significant sites are of value.

The Moore Haven site is located in the northwestern corner of Section 12, Township 42 South, Range 32 East, near the confluence of the Caloosahatchee River and Lake Okeechobee in the Town of Moore Haven, Florida. The site is located on a SFWMD-owned parcel of land adjacent to the S-77 spillway and lock, which conveys water to and from Lake Okeechobee's interior rim canal and the Caloosahatchee River. The proposed location is approximately 500 feet north of Highway 27, northwest of a housing community, and adjacent to the Caloosahatchee River (C-43 canal) at Latitude N 26° 50' 10", Longitude W 81° 05' 14". The site location is depicted on Figure 3-39. The site has been altered by construction and maintenance of the canal, bridge, access roads, and neighboring community. The majority of the site appears to be the site of discarded spoil material and construction debris, such as asphalt, concrete, etc. A dilapidated dock remains accessible on the shoreline. A natural forested area, most likely a forested wetland, borders the northeastern portion of inspected area. An access road divides the site. The western portion, adjacent to the canal, is used as a roadside park, complete with a boat ramp. The eastern portion is a grassy area, which has been used as a construction

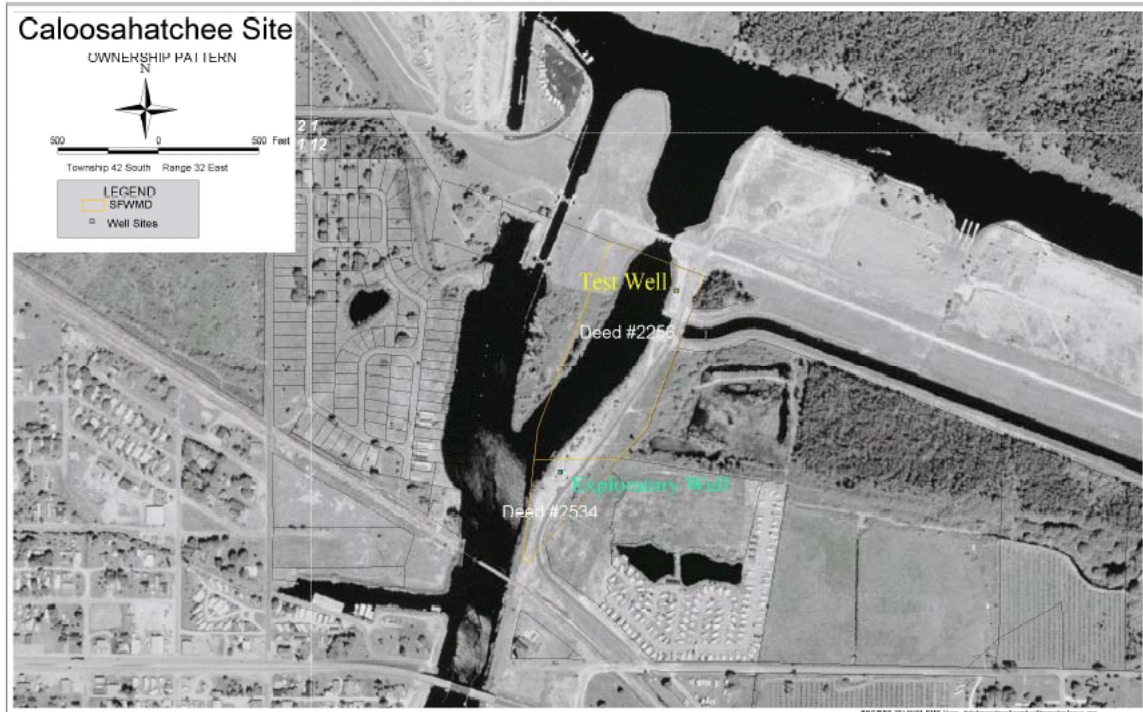


Figure 3-39. Location of the Moore Haven ASR pilot site (Source SFWMD).

site for a nearby bridge and dump. The site has been altered by past activities and is of poor quality for wildlife habitat. Tire tracks and discarded material indicate continual use of the area, possibly as parking lot for water access. Significant cultural resources are unlikely to be encountered on the site.

The Caloosahatchee River ASR Pilot Project site is located west of Lake Okeechobee, a few miles southwest of the City of LaBelle, just south of State Road 80, in western Hendry County, Florida. A project location map with alternative pilot project locations is presented on Figure 3-40 (Site # 1 on map). The proposed ASR facility is located in the northwestern quadrant of the northeastern corner of Section 6, Township 44 South, Range 28 East, on SFWMD-owned land referred to as “Berry Groves”. The coordinates of the exploratory well are Latitude: 26°41' 13.209"N, Longitude: 81° 33' 13.710"W.

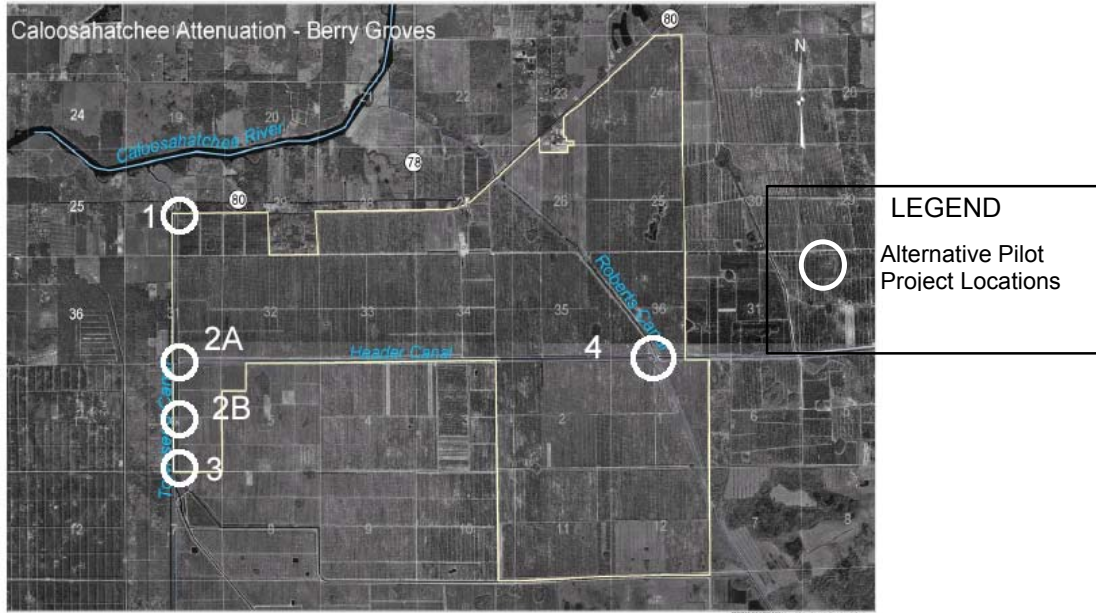


Figure 3-40. Location of the Caloosahatchee River ASR pilot site (Source SFWMD).

The planning decision framework developed as part of this research effort will be applied to the five ASR Pilot Project sites to determine how these sites may perform. The application of the framework may aid Corps of Engineers and SFWMD project planners by determining which of the five sites is the best or the worst as well as provide an estimate of economic feasibility.

Step 1. For the CERP the purpose of the proposed ASR wells is to aid with the storage, timing, and distribution of freshwater to the south Florida ecosystem. All of the proposed wells were proposed as high-capacity wells capable of storing and recovering vast quantities of freshwater. The plan assumed that all recovered freshwater would be returned to a surface water body within the existing surface water network. Since the recovered freshwater is to be utilized for ecosystem restoration, the water quality is of paramount importance.

Step 2. Each of the proposed pilot locations has excess source water available for storage in the Upper Floridan Aquifer System. Typically, the water is available during

the wet season, while during the dry season, water may be needed to meet irrigation, ecological, or potable demands. Since the CERP ASR recovered water is to be returned to the surface water network, any of the demand types could be satisfied. The surface water quality is quite variable and contains high levels of TSS, TOC, TDS, and chloride. The mean chloride levels were shown previously in Table 3-34. The level of TSS varies from one to ten mg/l at Hillsboro Canal to greater than 20 mg/l at the Port Mayaca and Moore Haven sites. TOC levels show similar patterns. At each site, a suitable aquifer storage zone has been found, although the aquifer transmissivities are on the low end of a regional range determined by others.

Geochemical issues are not expected for Kissimmee River, Hillsboro Canal, or Port Mayaca, however, problems may arise at Caloosahatchee River and Moore Haven. For those two sites, the upper portions of the Floridan Aquifer System may contain elevated concentrations of heavy metals including arsenic, manganese, uranium, nickel, and cobalt. As has been discussed in Chapter 2 of this report, the Lee County Olga ASR site has observed arsenic within the recovered water. This site is approximately 10 miles west of the Caloosahatchee River site. Therefore, this site may have the highest risk of geochemical problems.

Regulatory requirements in Florida require that all recharge water meet Class I drinking water standards upon injection into the Upper Floridan Aquifer, while also meeting Class III surface standards upon recovery. The most stringent Class I requirement during recharge calls for removal of 99% of pathogens such as coliforms. During recovery, it is anticipated that chloride, sulfate, specific conductance, TDS, and various metals will be recovered. Generally, metals will be recovered at concentrations

below the regulatory levels unless geochemical reactions occur in-situ as has happened in some ASR sites discussed in Chapter 2. It is expected that chloride will provide a good indicator of recovered water quality since it can be closely correlated with both TDS and specific conductance. Therefore, performance calculations of RE percentage will also use chloride as an indicator parameter.

Environmental constraints include temperature of the recovered water, potential ecological toxicity of the recovered water, and temporary disturbance of wading birds during pump operations due to noise and vibrations. Of the five pilot project sites, the Kissimmee River is located in the most sensitive ecological habitat. It is located in the floodplain of the river near freshwater marshes and fish spawning areas. Due to the possible presence of juvenile fish, special precautions are required for the design of the Kissimmee River pilot site surface water intake. In addition to the surface water intake, other site infrastructure required includes an ASR well, monitoring wells, water treatment plant, control building, piping, and a cascade aeration discharge structure.

As part of the initial site selection of the pilot project locations, an extensive site selection evaluation was completed. Site selection factors considered environmental impacts, groundwater quality, source water quality, social effects, availability of power, and others. This analysis has been previously documented by Brown et al. (2005c).

Step 3. As was discussed above, each pilot project site is being evaluated as a separate alternative for this report. All five sites will store 150,000,000 gallons of freshwater during a 30 day recharge period and will recover for up to 30 days. The water will be stored during the wet season and then recovered during the dry season. The water will be utilized initially for in-stream purposes, however, in the future it could be used for

potable or irrigation supply. Due to the source water quality, all of the sites will have water treatment plants that include filtration followed by disinfection. Filtration to TSS levels below 10 mg/l is recommended in this report to minimize the potential for well clogging for wells completed in limestone geology. For the Hillsboro and Caloosahatchee River sites, screen filters or underdrain filters will be utilized since the influent TSS values are generally less than 10 mg/l most of the time. Moore Haven, Port Mayaca, and the Kissimmee River sites will require more substantial pre-treatment prior to disinfection due to the higher TSS levels present in the surface water at each site. In order to minimize issues with disinfection by-products from chemical disinfection, each site will utilize UV disinfection instead. UV disinfection should be able to inactivate the pathogens while not generating any DBPs.

The costs are estimated to range from \$1.00 to \$2.00 per thousand gallons freshwater recovered, while the benefits will include water quality improvements, improvement in timing and distribution of water, and net water supply gain to the system since much of the water is currently flushed to tide to the Atlantic Ocean (USACE and SFWMD, 1999). The construction of the ASR pilot projects is scheduled to begin in early 2006.

Step 4. Now that all of the basic information has been compiled, each alternative plan can be evaluated to determine the effects on the aquifer, surface water bodies, and environment. Figure 3-41 depicts the estimated head changes within the Upper Floridan Aquifer System as determined by the Theis analytical model discussed previously.

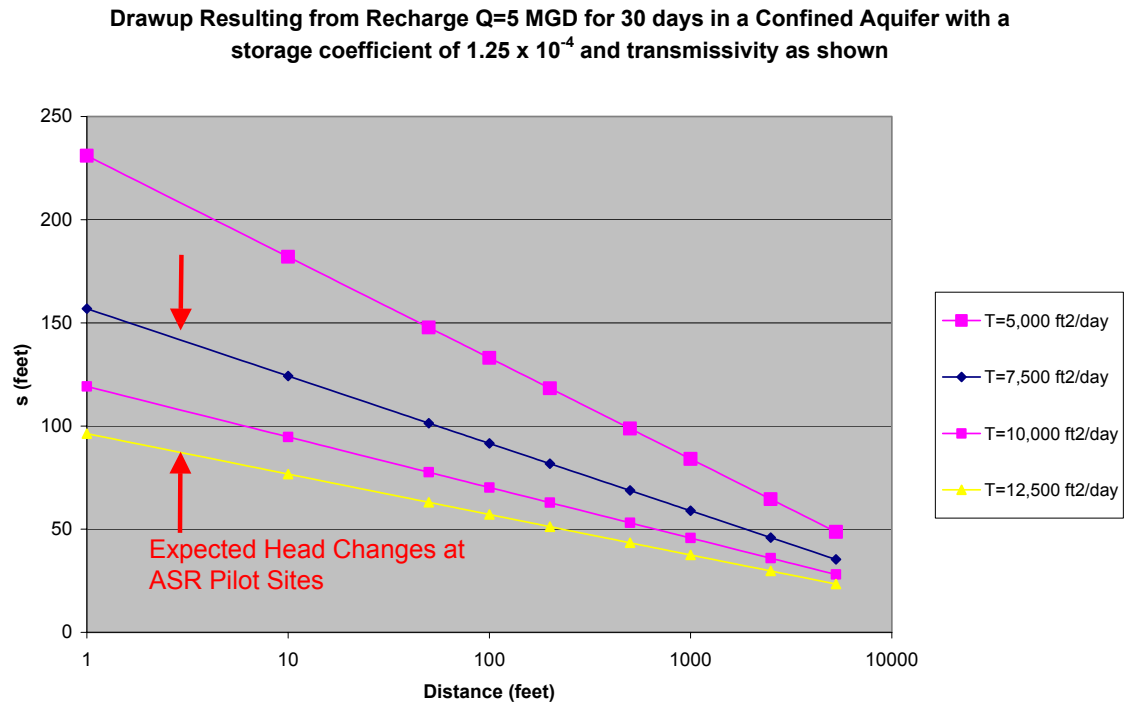


Figure 3-41. Estimated head changes within Upper Floridan Aquifer System at the ASR Pilot Projects

During recharge, water quality changes within the aquifer due to mixing and hydrodynamic dispersion are expected. The recovered water will be a blend of the recharge water and the ambient groundwater. Since the storage zones range in salinity from slightly brackish to moderately brackish, it is expected that the performance of the ASR pilot project will be variable for each site. In addition, the recovery of sulfate and sodium will need to be monitored in addition to TDS and chloride. Sodium can be toxic to many agricultural crops at concentrations over 250 mg/l. Furthermore, the recovered water will contain low to moderate levels of some emerging contaminants. The concentration of various emerging contaminants is expected to be below the levels found within the recharge surface water body due to mixing and hydrodynamic dispersion in the aquifer. The same should be true for nutrients so that a net water quality improvement for these categories is expected. No acute or chronic toxicity effects are expected related

to ecological receptors (Golder and Associates, 2005), however, toxicity testing is recommended should a pilot project be completed. Temperature changes within the surface water are expected and will have to be mitigated using a mixing zone.

Temperature changes in each surface water body due to recovery of warmer stored ASR water could be problematic during extreme drought events. More detailed engineering evaluations should be considered during any recommended pilot project.

Step 5. In comparing the five alternatives developed for ASR Pilot Projects, it is important to estimate the ASR RE percentage and long-term CRE percentage for each. In order to develop those estimates, the recovery index and inverse relative dispersivity dimensionless parameters will be calculated for each alternative. For the five alternatives, all information necessary to calculate the dimensionless parameters is available except the dispersivity. As discussed in this report, two approaches are recommended. First, a suggested planning guideline value of 30 feet for the dispersivity can be used for the Hillsboro, Port Mayaca, Kissimmee River and Moore Haven sites since the geologic character of each is aptly described as limestone with minor conduit features. For the Caloosahatchee River site, a guideline value of 7.5 feet is recommended for the dispersivity since the geologic character can be described as a sandy limestone.

Second, the dispersivity can be estimated based upon the geologic character and the scale of the project. Using equation 6 of this report and replacing “L” with r_m , a reasonable second estimate of dispersivity can be approximated at 47 feet for Hillsboro, Port Mayaca, Kissimmee River, and Moore Haven, while the power law model estimates a dispersivity value of 29.8 feet for the Caloosahatchee River site. Both of the estimated dispersivity values can then be used in the dimensionless parameters to develop estimates

of RE percentage. Both the linear regression model # 1 and the performance envelopes are available to assist in this task. Therefore, three separate RE estimates can be produced for each alternative. The results are then averaged to evaluate the CRE percentage.

Table 3-35 provides a summary of the recharge volumes for each alternative, the dimensionless parameters, and the estimated initial cycle one RE percentage. The first five estimates utilize the dispersivity values calculated from the power law equation 6. The second five estimates utilize the dispersivity guideline values of 30 feet and 7.5 feet, respectively.

Table 3-35. ASR Pilot Projects with predicted RE percentage.

Alt #	Recharge Volume (ft ³)	Aquifer thickness b (feet)	Porosity %	r _m (feet)	LN (R _α ')	LN (RI)	RE estimate %
Hillsboro	150,000,000	200	25	357.3	2.02	2.94	0.86
Port Mayaca	150,000,000	200	25	357.3	2.02	1.73	15.99
Kiss River	150,000,000	200	25	357.3	2.02	0.98	36.22
Moore Haven	150,000,000	200	25	357.3	2.02	2.77	0.76
Caloosahatchee River	150,000,000	200	25	357.3	2.48	1.39	35.54
Hillsboro	150,000,000	200	25	357.3	2.48	2.94	11.50
Port Mayaca	150,000,000	200	25	357.3	2.48	1.73	26.63
Kiss River	150,000,000	200	25	357.3	2.48	0.98	46.86
Moore Haven	150,000,000	200	25	357.3	2.48	2.77	11.40
Caloosahatchee River	150,000,000	200	25	357.3	3.86	1.39	57.96

As the table clearly demonstrates, the initial cycle one RE percentage is predicted to range from zero to 58% using both proposed planning approaches. Using the graphical method as presented in Figures 3-42 to 3-44, the initial cycle one RE estimates range from 11 to 43 percent.

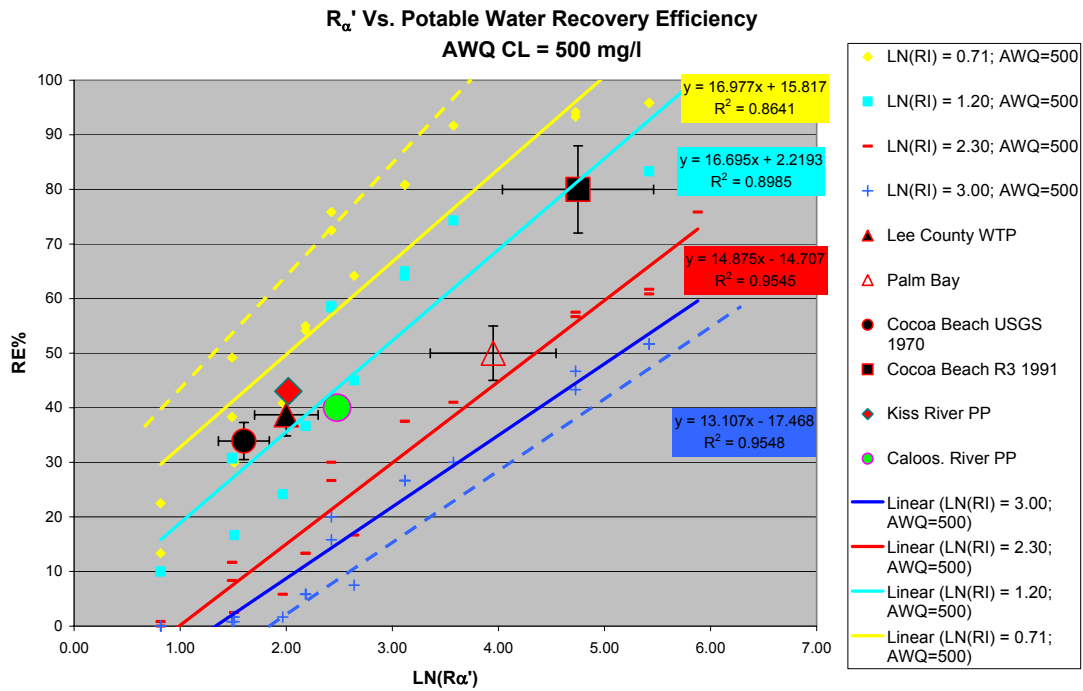


Figure 3-42. RE estimates for Kissimmee River and Caloosahatchee River ASR Pilot Projects.

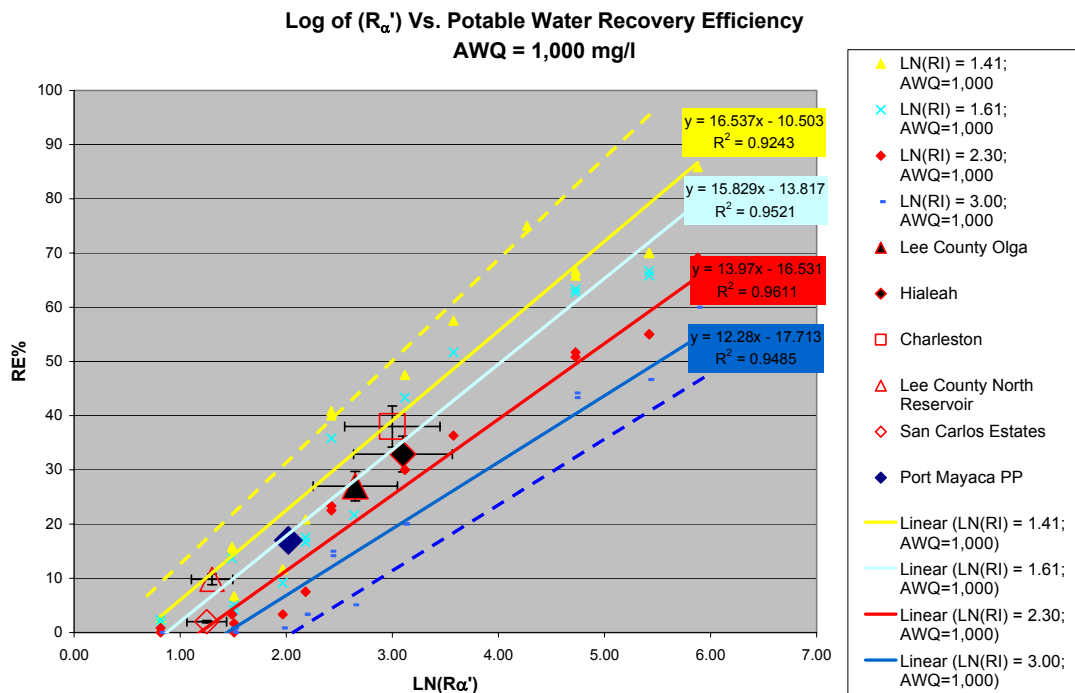


Figure 3-43. RE estimate for Port Mayaca ASR Pilot Project.

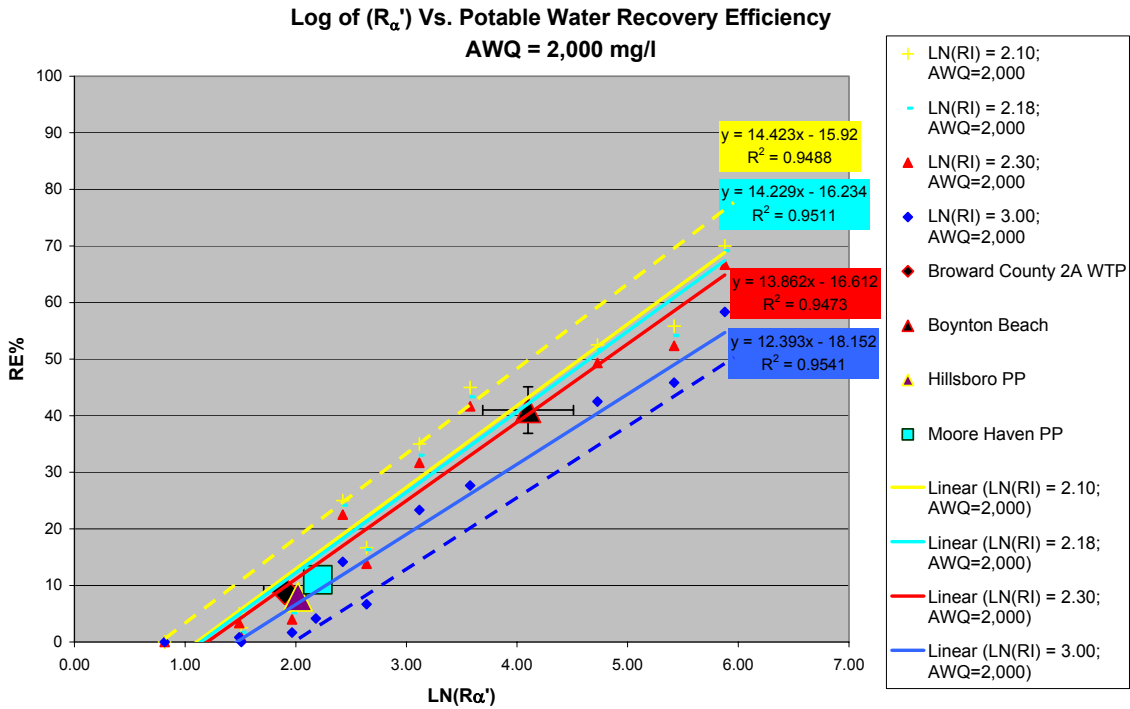


Figure 3-44. RE estimates for Hillsboro Canal and Moore Haven ASR Pilot Projects.

The means of the three RE estimates for each site are:

- Hillsboro Canal 6.67 %
- Port Mayaca 19.9%
- Kissimmee River 42.0%
- Moore Haven 8.05%
- Caloosahatchee River 44.5%

Now this value can be utilized in one of the CRE charts to estimate long-term performance. Figures 3-45 and 3-46 present the estimates for the ASR pilots. The figures clearly reveal the estimated CRE after six cycles for each pilot project. The CRE at the Hillsboro and Moore Haven sites approach 33% and 40%, respectively. The CRE after six cycles at the Port Mayaca, Kissimmee River, and Caloosahatchee River sites approach 60%, 77%, and 82%, respectively.

From the analysis, it appears that the Hillsboro site would approach an asymptote between 35 and 45% while Moore Haven would approach an asymptote of between 40 and 50%, similar to what has been observed at other existing ASR projects.

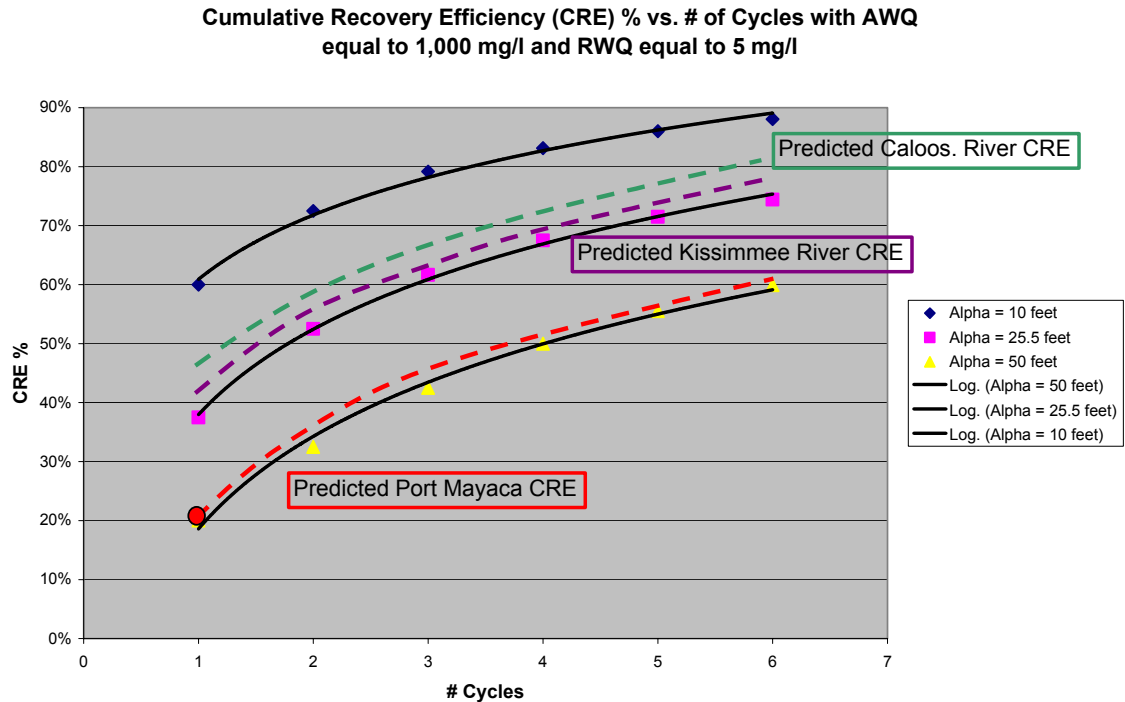


Figure 3-45. CRE estimates for Kissimmee River, Caloosahatchee River, and Port Mayaca ASR Pilot Projects.

Obviously, the poor recovery affects both the unit cost of the project and the benefits at the Hillsboro and Moore Haven sites. As has been pointed out previously, the unit cost can increase rapidly as the RE percentage or CRE percentage decreases. Similarly, the benefits also decrease substantially. Due to the low CRE values at Hillsboro and Moore Haven, the estimated unit cost would be \$2.00 to \$3.00 per thousand gallons water recovered at those sites. At this unit cost, other water resources options including reservoirs or wastewater reuse or just water treatment may be more economical. Since the main benefit of alternatives is water quality improvements to the

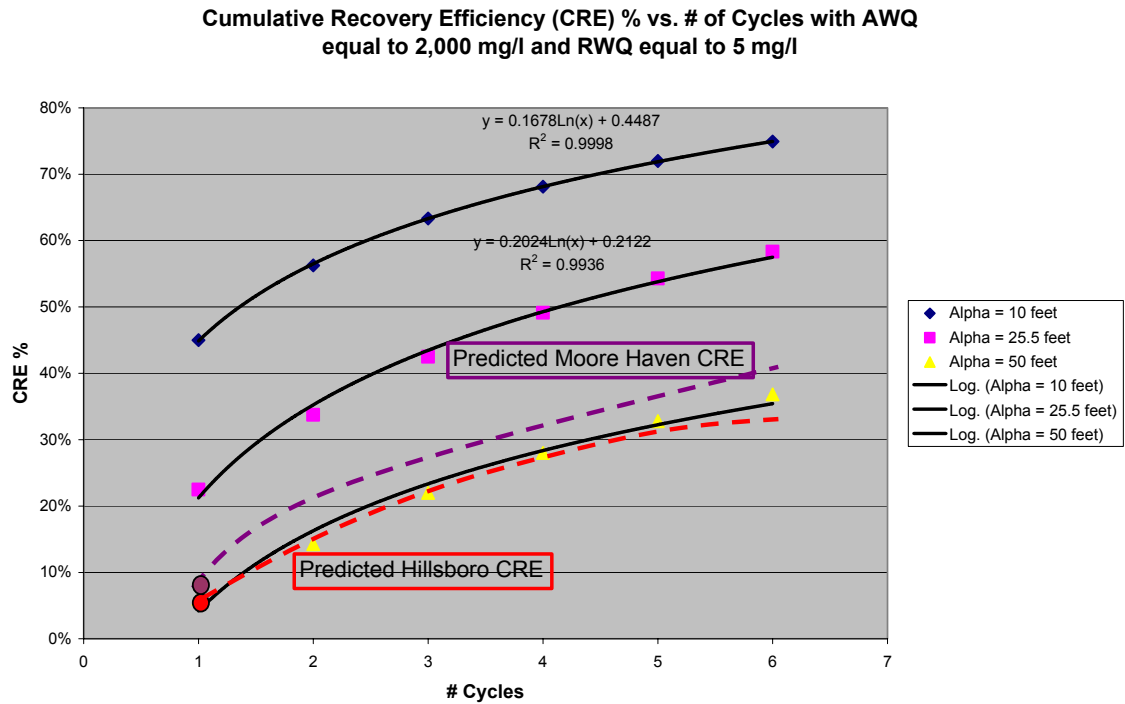


Figure 3-46. CRE estimates for Hillsboro Canal and Moore Haven ASR Pilot Projects. environment, these options may still provide limited positive contributions. The other three remaining pilot sites all seem quite economical and should cost between \$1.00 and \$2.00 per thousand gallons water recovered.

To complete the remaining alternative comparisons, the water quality of the recovered water for each alternative should be weighed for environmental impacts. The recovered water quality for irrigation use would not be problematic, however, the recovered water for in-stream use may add significant amounts of chloride, sulfate, and sodium to the surface water. As far as risk and uncertainties go, the main uncertainty is the dispersion that will actually occur, however, the analysis already used a reasonable range of dispersivity estimates. For economic risks, the main issue would be the possibility that the CRE percentage at the Hillsboro site is even lower than 33% after six

cycles. This is possible and would push the unit cost to more than \$3.75 per thousand gallons water recovered.

Based upon application of the newly proposed ASR planning decision framework, the five ASR pilot projects would provide a wide range of benefits and unit costs. The Caloosahatchee River, Kissimmee River, and Port Mayaca sites seem quite promising and should continue on to pilot testing. The Moore Haven site seems to provide fair performance and probably deserves to move to the pilot project stage in order to optimize the performance. Further optimization of the ASR storage zone selection at the Moore Haven site probably will result in much improved recovery efficiency. In essence, selection of a shallower and well confined storage zone at this relocation should result in improved system performance.

The Hillsboro Canal site performs the worst based upon the framework evaluations. There are a number of reasons for the projected poor performance including high dispersivity, poor ambient water quality, and poor source water quality. This site may not be the best candidate for pilot testing, however, the project has been approved for construction in 2006. Therefore, pilot testing is scheduled to commence in early 2006. The USACE and the SFWMD hope that pilot testing may provide new methods and new information to improve the expected system performance. In addition, alternative uses of the recovered water from the Hillsboro site should be considered. Since the Hillsboro site is located in a highly urbanized environment, potable water supply or irrigation water supply alternatives should also be investigated. Both potable water use or irrigation use are expected to result in increased recovery of stored water since the regulatory requirements would actually be reduced as compared to use for environmental

restoration. Alternatively, the USACE and SFWMD could seek additional water quality criteria exemptions in order to reduce the overall water treatment cost. In this manner, the unit cost could be reduced.

CHAPTER 4

CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER RESEARCH

Artificial recharge provides a means to enhance natural groundwater recharge, thereby improving an aquifer's ability to provide a dependable water supply for human and ecologic demands. Artificial recharge has been utilized extensively around the world through the application of both surface spreading methods and injection wells. Aquifer, storage and recovery (ASR) is a sub-set of artificial recharge where water is stored when available and then recovered from the same well when it is needed. ASR has many advantages including a lack of permanence, limited water losses due to evaporation, and low vulnerability to intentional destruction or contamination. ASR has primarily been used to meet seasonal water supply demands but other uses such as improving water quality, restoring water levels, and reducing subsidence have been documented.

Regulation of ASR is governed by multiple Federal and State regulatory regimes resulting in a complicated legal framework involving a myriad of statutes. In the United States, ASR technology often proceeds in advance of regulatory guidelines leading to convoluted and inconsistent requirements and constraints. This research effort has focused upon the development of a new ASR planning decision framework. The adoption of this proposed planning framework would empower regulators across Florida and the United States to develop a more consistent regulatory framework that promotes more technological development of ASR instead of constraining it. In addition, adoption of this framework will likely improve the overall protection of human and ecological water users that are dependent upon ASR projects.

There are four primary constraints to ASR project development including:

- Regulatory
- Recharge and recovered water quality
- Water availability and demand
- Availability of a suitable aquifer

Of the constraints listed, the overall water quality of the recharge and recovered water is most important. The importance of water quality in ASR projects is embedded in the long list of physical or chemical compounds that must be considered during project planning. Generally, five different water quality categories must be thoroughly examined to determine the potential project feasibility, including:

- Conventional parameters (e.g., TSS, pH, TOC)
- Toxics
- Pathogens
- Disinfection By-products
- Emerging Contaminants

The water quality of the recharge and recovered water will control the applicable uses of the ASR project. Potable ASR projects require water that meets the most stringent Federal and State regulatory requirements, while irrigation and in-stream beneficial water use may be permitted using water with slightly sub-standard water quality.

Planning of past ASR projects has varied considerably and has not been consistent. No standardized methodology has been available to guide the prospective project development. The level of detail afforded different planning factors varies between public and private ASR proponents depending upon public or agency interest, financial resources, and available schedules. Improved planning methodologies are required for both non-brackish and brackish water ASR projects, with the later requiring more careful deliberations.

This research report has thoroughly reviewed the planning methodology for ASR projects. The review was comprehensive in nature and included available literature as well as ASR case study information for 50 different ASR projects from around the globe. Relevant literature from the 1960s to the present time has been summarized and contrasted to determine the most important ASR performance factors. An in-depth evaluation and comparison of numerical model simulations of various ASR performance factors was also performed for this research report. This comparison is the first of its kind to be developed to discern important theoretical aspects of ASR performance. The performance comparison marks an important contribution to the planning and design of future ASR projects in Florida and around the world. The compilation of case study information of 50 different ASR projects is thought to be the most comprehensive comparison of ASR sites that has been completed to date. This information should be useful to future ASR researchers from Florida and around the world, as a source of real operational performance data and “lessons learned”. Based upon the model simulations and the case study information, key ASR performance factors include the aquifer hydrogeological characteristics, operational considerations, water quality, geochemistry, potential environmental impacts, required water treatment, and unit costs.

The research effort also distilled the most critical ASR performance factors that need to be considered, their required order of completion, and the technical studies needed, to complete a thorough planning effort. The research determined that the most important performance variables involve hydrodynamic dispersion, water quality, and recharge volume. Based upon the literature, the newly developed ASR case histories, and numerical modeling simulations, a comprehensive planning decision framework has been

constructed. The planning decision framework is a first of its kind standardized planning methodology for ASR projects and represents an original contribution to the science of ASR planning in Florida, the United States, and around the world. It should be invaluable to anyone investigating the feasibility of new ASR projects as a viable water resources option. The decision framework is thought to be applicable to any ASR project site but it has been tailored to support more complicated planning efforts required at brackish water ASR sites.

In support of the planning decision framework, new tools to predict ASR recovery efficiency at brackish water ASR projects have been developed. These innovative tools include both multivariate regression models and graphical planning methods. Some of the graphical planning aids were developed based upon published analytical models, while others represent completely original research contributions. All of the original tools are based upon the use of dimensionless parameters that combine the most critical ASR performance variables mentioned above into simple ratios. Two prominent dimensionless parameters are proposed for use including the inverse relative dispersivity and the recovery index. The inverse relative dispersivity is a revision of a previously developed dimensionless parameter while the recovery index is a novel parameter developed specifically for this research report.

Generally, all information required for input into the dimensionless parameters is readily available with the exception of estimates of dispersivity. Therefore, two innovative technical approaches are offered to aid with the development of reasonable dispersivity estimates. First, guidance values based upon the predominant geologic character of the aquifer storage zone are provided. Second, an adaptation of another

recently published dispersivity estimating method is proposed for consideration and use. The adapted estimating method should prove valuable to ASR practitioners and represents another original contribution to ASR science.

After the development of all of the new guidance values, performance metrics, and planning decision framework, testing the framework at two ASR sites from around the United States completed validation testing of the new planning methodology. The first site was the brackish water Lake Okeechobee Taylor Creek site in south Florida, a site that had been unsuccessful in the late 1980s. The new planning methodology was employed and revealed that the Taylor Creek site would not likely be very successful or economically feasible. This conclusion agrees with the actual observed experience at the Taylor Creek site. The second site tested was the Kerrville site in Texas. This site is a non-brackish water project site considered highly successful by utility owners. The new planning methodology was applied and concluded that the Kerrville project was feasible and should be highly successful, matching the actual project observations.

The final demonstration and illustration of the ASR planning decision framework was the application of the new planning methodology at five proposed ASR pilot projects proposed as part of the Comprehensive Everglades Restoration Project (CERP). The methodology was applied to the five sites to determine the sites likely to have the best and worst performance. The planning framework revealed that the Caloosahatchee River and Kissimmee River pilot projects should perform the best, while the Hillsboro Canal pilot project would probably result in the worst performance.

In conclusion, the body of the planning decision framework is built upon existing information, some of which has never been summarized or published before. The

internal workings or organs of the framework have been built through the development of new ASR planning guidance and completely original performance metrics. Extensive numerical simulation models support much of the proposed guidance tables and new performance metrics; these are the blood of the decision framework. Lastly, the skin of the framework that holds it all together is the development of new planning procedures and logic. The framework logic is presented in a series of flow charts and figures that allow easy use by ASR practitioners. It is thought that the use and application of the final ASR planning decision framework will greatly improve feasibility level planning activities in the future and should result in less failed projects, thus saving considerable time and resources for the economies of the Florida and United States.

During the development of this research report, several issues have surfaced that probably warrant additional research. First, the addition of supplementary ASR case study information to the database of sites established in this report would likely further improve ASR planning through the gathering of additional “lessons learned”. Summaries of other existing failed ASR projects would be especially important.

Second, conclusions drawn from numerical modeling completed for this research report conflict with past literature in several instances related to three ASR performance factors including:

- Transmissivity
- Porosity
- Heterogeneity

Additional numerical modeling of these performance factors is recommended to determine the extent of each variable’s importance.

Third, the dimensionless parameters developed for this research report to aid with estimating the initial site recovery efficiency are limited to brackish water sites where the ambient groundwater chloride concentration is less than 2,500 mg/l or TDS is less than 5,000 mg/l. At more saline sites, the dimensionless parameters may still have some validity or they may have to be modified to consider density effects. Incorporating density into the dimensionless parameters themselves may also be possible, leading to further refinements and improvements.

Lastly, the scale dependent nature of dispersivity should continue to be investigated as it relates to ASR performance. One of the dimensionless parameters utilized in this report, the inverse relative dispersivity, is a simple ratio between the project scale and the dispersivity. For this report, the assumption was made that the dispersivity has a weak correlation to the project scale rather than being entirely dependent upon it. This assumption may not be valid at large project scales where the dispersivity may be more dependent upon scale length than the geologic character of the material. Additional ASR project site data should be reviewed to refine the proposed dimensionless parameters with respect to the effect of project scale. Also, model simulations completed in this report and past efforts discussed within the literature assume constant dispersivity values. This conceptual model may be incorrect as the scale of ASR projects increases. A new conceptual model that incorporates the scale of the recharged “bubble” as a consideration in assigning dispersivity values in numerical models is probably warranted. This new conceptual model would also affect the planned future operation of some larger ASR projects, in that, the so called target storage volume (TSV) approach may actually be counter productive beyond some critical injection

volume. Further numerical modeling could employ the adapted power law model discussed in Chapter 3 of this report to estimate dispersivities across large model scales. Using this new conceptual model of dispersion, additional model runs could determine the recharge volume size that provides the most benefit to initial recovery efficiency.

APPENDIX A
APPENDIX A - LIST OF PUBLICATIONS DERIVED FROM DISSERTATION

This report has resulted in a number of publications including portions of two reports prepared by the U.S. Army Corps of Engineers, three peer-reviewed journal articles, and two peer-reviewed conference proceedings. Hopefully, this research effort will result in additional publications and collaborations. The publications include:

- U.S. Army Corps of Engineers (USACE) and the South Florida Water Management District (SFWMD), 2004. Final Pilot Project Design Report/Environmental Impact Statement, Lake Okeechobee ASR Pilot Project, Hillsboro ASR Pilot Project and Caloosahatchee River ASR Pilot Project, U.S. Army Corps of Engineers, Jacksonville, Florida, September 2004, 400 p. (portions of this report)
- Brown, C., Itani, S., & Zhang, M., 2004. Final Draft Report, ASR Regional Study: A Scientific Evaluation of Potential Pressure Induced Constraints and Changes in the Floridan Aquifer System and the Hawthorn Group, U.S. Army Corps of Engineers, Jacksonville, Florida, 64 p. (portions of this report)
- Brown, C., E. Brown, and S. Sutterfield, 2005a. Considerations for the CERP ASR Contingency Plan. *Florida Water Resources Journal*, February 2005:32-33.
- Brown, C., Weiss, R., Verrastro, R., and Schubert, S., 2005b. Development of an Aquifer, Storage and Recovery (ASR) Site Selection Suitability Index in Support of the Comprehensive Everglades Restoration Project. *Journal of Environmental Hydrology*, 13(20):1-13.
- Brown, C., S. Sutterfield, J. Hendel, P. Kwiatkowski, J. Mirecki, and K. Hatfield, 2005c. The Comprehensive Everglades Restoration Program ASR Pilot Projects – Plan Development. *Journal of Water Resources Planning and Management, ASCE*, submitted May 2005, 20 p.
- Brown, C. and R. Nevulis, 2005. A Model Study of the Proposed Everglades Hillsboro ASR Pilot Project. International Symposium on Artificial Recharge 5, American Society of Civil Engineers and the United Nations (Editors), accepted August 2005, 7 p.

Brown, C. and L. Motz, 2005. A Comparison of the Operational Performance at Ten ASR Sites located within the United States, Australia, and England. International Symposium on Artificial Recharge 5, American Society of Civil Engineers and the United Nations (Editors), accepted August 2005, 6 p.

APPENDIX B PARTIAL COMPILATION OF ASR MODEL RESULTS

This report has included extensive numerical modeling. A partial summary of results is presented within this appendix. Figure B-1 depicts summary tables extracted from Microsoft Excel ®.

AWQ	Q (MGD)	Vol in ft^3	Alpha L	LN Alpha	ADV Rad	Adv Rad/Alpha	LN (rm/alpha)	T	RWQ	RI	LN (RI)	REFF%
500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	5.00	2.04	0.71	13.33
500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	25000.00	5.00	2.04	0.71	22.50
500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	5.00	2.04	0.71	30.00
500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	5.00	2.04	0.71	30.00
500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	5.00	2.04	0.71	38.33
500.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	5.00	2.04	0.71	40.83
500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	25000.00	5.00	2.04	0.71	49.17
500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	5.00	2.04	0.71	54.17
500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	5.00	2.04	0.71	55.00
500.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	5.00	2.04	0.71	64.17
500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	5.00	2.04	0.71	72.50
500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	25000.00	5.00	2.04	0.71	75.83
500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	5.00	2.04	0.71	80.83
500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	5.00	2.04	0.71	80.83
500.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	5.00	2.04	0.71	91.67
500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	25000.00	5.00	2.04	0.71	93.33
500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	5.00	2.04	0.71	94.17
500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	5.00	2.04	0.71	95.83
500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	5.00	2.04	0.71	95.83
500.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	5.00	2.04	0.71	113.00
500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	25000.00	100.00	3.33	1.20	10.00
500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	100.00	3.33	1.20	10.00
500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	100.00	3.33	1.20	16.67
500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	100.00	3.33	1.20	16.67
500.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	100.00	3.33	1.20	24.17
500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	100.00	3.33	1.20	30.67
500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	25000.00	100.00	3.33	1.20	30.83
500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	100.00	3.33	1.20	36.67
500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	100.00	3.33	1.20	36.67
500.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	100.00	3.33	1.20	45.00
500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	25000.00	100.00	3.33	1.20	58.33
500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	100.00	3.33	1.20	58.67
500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	100.00	3.33	1.20	64.17
500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	100.00	3.33	1.20	65.00
500.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	100.00	3.33	1.20	74.33
500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	100.00	3.33	1.20	78.67
500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	25000.00	100.00	3.33	1.20	80.00
500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	100.00	3.33	1.20	83.33
500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	100.00	3.33	1.20	83.33
500.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	100.00	3.33	1.20	100.00
1000.00	0.50	2005347.59	50.00	3.91	112.99	2.26	0.82	10000.00	5.00	4.08	1.41	1.25
1000.00	0.50	2005347.59	50.00	3.91	159.79	3.20	1.16	10000.00	5.00	4.08	1.41	2.50
1000.00	0.50	2005347.59	50.00	3.91	112.99	2.26	0.82	10000.00	5.00	4.08	1.41	2.50
1000.00	0.50	2005347.59	50.00	3.91	159.79	3.20	1.16	5000.00	5.00	4.08	1.41	2.50
1000.00	0.50	2005347.59	50.00	3.91	112.99	2.26	0.82	10000.00	5.00	4.08	1.41	2.50
1000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	25000.00	5.00	4.08	1.41	2.50
1000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	5.00	4.08	1.41	2.50
1000.00	0.50	668449.20	50.00	3.91	92.25	1.85	0.61	250000.00	5.00	4.08	1.41	3.75
1000.00	0.50	668449.20	50.00	3.91	92.25	1.85	0.61	250000.00	5.00	4.08	1.41	3.75
1000.00	0.50	668449.20	50.00	3.91	92.25	1.85	0.61	250000.00	5.00	4.08	1.41	3.75
1000.00	0.50	668449.20	50.00	3.91	92.25	1.85	0.61	10000.00	5.00	4.08	1.41	3.75
1000.00	0.50	668449.20	50.00	3.91	119.10	2.38	0.87	10000.00	5.00	4.08	1.41	3.75
1000.00	0.50	668449.20	50.00	3.91	65.23	1.30	0.27	10000.00	5.00	4.08	1.41	3.75
1000.00	0.50	2005347.59	50.00	3.91	159.79	3.20	1.16	5000.00	5.00	4.08	1.41	5.00
1000.00	0.50	2005347.59	50.00	3.91	159.79	3.20	1.16	5000.00	5.00	4.08	1.41	5.00
1000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	5.00	4.08	1.41	6.67
1000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	5.00	4.08	1.41	6.67
1000.00	0.50	2005347.59	25.50	3.24	112.99	4.43	1.49	10000.00	5.00	4.08	1.41	7.50
1000.00	0.50	2005347.59	25.50	3.24	159.79	6.27	1.84	10000.00	5.00	4.08	1.41	9.10
1000.00	0.50	2005347.59	25.50	3.24	112.99	4.43	1.49	10000.00	5.00	4.08	1.41	10.00
1000.00	0.50	668449.20	50.00	3.91	130.47	2.61	0.96	2500.00	5.00	4.08	1.41	10.00
1000.00	0.50	668449.20	50.00	3.91	130.47	2.61	0.96	25000.00	5.00	4.08	1.41	10.00
1000.00	0.50	668449.20	50.00	3.91	130.47	2.61	0.96	250000.00	5.00	4.08	1.41	10.00
1000.00	0.50	2005347.59	25.50	3.24	159.79	6.27	1.84	5000.00	5.00	4.08	1.41	10.83
1000.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	5.00	4.08	1.41	11.67
1000.00	0.50	2005347.59	25.50	3.24	159.79	6.27	1.84	5000.00	5.00	4.08	1.41	12.50
1000.00	0.50	2005347.59	25.50	3.24	159.79	6.27	1.84	5000.00	5.00	4.08	1.41	15.00
1000.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	25000.00	5.00	4.08	1.41	15.00
1000.00	0.50	2005347.59	25.50	3.24	112.99	4.43	1.49	10000.00	5.00	4.08	1.41	15.83
1000.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	5.00	4.08	1.41	15.83
1000.00	0.50	668449.20	25.50	3.24	92.25	3.62	1.29	250000.00	5.00	4.08	1.41	18.75
1000.00	0.50	668449.20	25.50	3.24	92.25	3.62	1.29	250000.00	5.00	4.08	1.41	18.75

1000.00	0.50	668449.20	25.50	3.24	92.25	3.62	1.29	250000.00	5.00	4.08	1.41	18.75
1000.00	0.50	668449.20	25.50	3.24	92.25	3.62	1.29	10000.00	5.00	4.08	1.41	18.75
1000.00	0.50	668449.20	25.50	3.24	119.10	4.67	1.54	10000.00	5.00	4.08	1.41	18.75
1000.00	0.50	668449.20	25.50	3.24	65.23	2.56	0.94	10000.00	5.00	4.08	1.41	18.75
1000.00	0.50	668449.20	25.50	3.24	92.25	3.62	1.29	10000.00	5.00	4.08	1.41	18.75
1000.00	0.50	2005347.59	25.50	3.24	159.79	6.27	1.84	5000.00	5.00	4.08	1.41	20.00
1000.00	4.00	2673796.79	50.00	3.91	184.51	3.69	1.31	10000.00	5.00	4.08	1.41	20.00
1000.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	5.00	4.08	1.41	20.83
1000.00	0.50	2005347.59	10.00	2.30	112.99	11.30	2.42	10000.00	5.00	4.08	1.41	25.00
1000.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	5.00	4.08	1.41	25.83
1000.00	0.50	668449.20	25.50	3.24	130.47	5.12	1.63	2500.00	5.00	4.08	1.41	27.50
1000.00	0.50	668449.20	25.50	3.24	130.47	5.12	1.63	25000.00	5.00	4.08	1.41	27.50
1000.00	0.50	668449.20	25.50	3.24	130.47	5.12	1.63	250000.00	5.00	4.08	1.41	27.50
1000.00	1.00	1336898.40	25.50	3.24	130.47	5.12	1.63	10000.00	5.00	4.08	1.41	27.75
1000.00	0.50	2005347.59	10.00	2.30	159.79	15.98	2.77	10000.00	5.00	4.08	1.41	30.00
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	10000.00	5.00	4.08	1.41	31.25
1000.00	0.50	668449.20	10.00	2.30	119.10	11.91	2.48	10000.00	5.00	4.08	1.41	31.25
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	10000.00	5.00	4.08	1.41	31.25
1000.00	0.50	2005347.59	10.00	2.30	159.79	15.98	2.77	5000.00	5.00	4.08	1.41	32.50
1000.00	0.50	2005347.59	10.00	2.30	159.79	15.98	2.77	5000.00	5.00	4.08	1.41	32.50
1000.00	0.50	2005347.59	10.00	2.30	112.99	11.30	2.42	10000.00	5.00	4.08	1.41	35.00
1000.00	2.00	2673796.79	25.50	3.24	184.51	7.24	1.98	10000.00	5.00	4.08	1.41	37.50
1000.00	2.00	2673796.79	25.50	3.24	184.51	7.24	1.98	10000.00	5.00	4.08	1.41	37.50
1000.00	1.00	2673796.79	25.50	3.24	184.51	7.24	1.98	10000.00	5.00	4.08	1.41	37.50
1000.00	4.00	2673796.79	25.50	3.24	184.51	7.24	1.98	10000.00	5.00	4.08	1.41	37.50
1000.00	4.00	2673796.79	25.50	3.24	184.51	7.24	1.98	10000.00	5.00	4.08	1.41	37.50
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	10000.00	5.00	4.08	1.41	40.00
1000.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	25000.00	5.00	4.08	1.41	40.00
1000.00	0.50	2005347.59	10.00	2.30	112.99	11.30	2.42	10000.00	5.00	4.08	1.41	40.83
1000.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	5.00	4.08	1.41	40.83
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	250000.00	5.00	4.08	1.41	43.75
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	250000.00	5.00	4.08	1.41	43.75
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	250000.00	5.00	4.08	1.41	43.75
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	10000.00	5.00	4.08	1.41	43.75
1000.00	0.50	668449.20	10.00	2.30	119.10	11.91	2.48	10000.00	5.00	4.08	1.41	43.75
1000.00	0.50	668449.20	10.00	2.30	65.23	6.52	1.88	10000.00	5.00	4.08	1.41	43.75
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	10000.00	5.00	4.08	1.41	43.75
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	10000.00	5.00	4.08	1.41	43.75
1000.00	0.50	668449.20	10.00	2.30	92.25	9.23	2.22	10000.00	5.00	4.08	1.41	43.75
1000.00	0.50	2005347.59	10.00	2.30	159.79	15.98	2.77	5000.00	5.00	4.08	1.41	45.00
1000.00	0.50	2005347.59	1.00	0.00	112.99	112.99	4.73	10000.00	5.00	4.08	1.41	47.50
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	10000.00	5.00	4.08	1.41	47.50
1000.00	4.00	5347593.58	25.50	3.24	260.94	10.23	2.33	10000.00	5.00	4.08	1.41	47.50
1000.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	5.00	4.08	1.41	47.50
1000.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	5.00	4.08	1.41	47.50
1000.00	0.50	2005347.59	10.00	2.30	159.79	15.98	2.77	5000.00	5.00	4.08	1.41	50.50
1000.00	0.50	668449.20	1.00	0.00	65.23	65.23	4.18	10000.00	5.00	4.08	1.41	51.25
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	10000.00	5.00	4.08	1.41	51.25
1000.00	0.50	668449.20	1.00	0.00	119.10	119.10	4.78	10000.00	5.00	4.08	1.41	51.25
1000.00	0.50	2005347.59	1.00	0.00	159.79	159.79	5.07	5000.00	5.00	4.08	1.41	52.25
1000.00	0.50	668449.20	10.00	2.30	130.47	13.05	2.57	2500.00	5.00	4.08	1.41	52.50
1000.00	0.50	668449.20	10.00	2.30	130.47	13.05	2.57	25000.00	5.00	4.08	1.41	52.50
1000.00	0.50	668449.20	10.00	2.30	130.47	13.05	2.57	250000.00	5.00	4.08	1.41	52.50
1000.00	1.00	1336898.40	10.00	2.30	130.47	13.05	2.57	10000.00	5.00	4.08	1.41	52.50
1000.00	8.00	10695187.17	25.50	3.24	369.02	14.47	2.67	10000.00	5.00	4.08	1.41	55.00
1000.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	5.00	4.08	1.41	57.50
1000.00	1.00	2673796.79	10.00	2.30	184.51	18.45	2.92	10000.00	5.00	4.08	1.41	59.38
1000.00	2.00	2673796.79	10.00	2.30	184.51	18.45	2.92	10000.00	5.00	4.08	1.41	60.00
1000.00	2.00	2673796.79	10.00	2.30	184.51	18.45	2.92	10000.00	5.00	4.08	1.41	60.00
1000.00	4.00	2673796.79	10.00	2.30	184.51	18.45	2.92	10000.00	5.00	4.08	1.41	60.00
1000.00	4.00	2673796.79	10.00	2.30	184.51	18.45	2.92	10000.00	5.00	4.08	1.41	60.00
1000.00	0.50	2005347.59	1.00	0.00	159.79	159.79	5.07	5000.00	5.00	4.08	1.41	65.00
1000.00	0.50	2005347.59	1.00	0.00	159.79	159.79	5.07	5000.00	5.00	4.08	1.41	65.00
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	10000.00	5.00	4.08	1.41	65.00
1000.00	8.00	39572192.51	25.50	3.24	709.82	27.84	3.33	10000.00	5.00	4.08	1.41	65.54
1000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	25000.00	5.00	4.08	1.41	65.83
1000.00	4.00	5347593.58	10.00	2.30	260.94	26.09	3.26	10000.00	5.00	4.08	1.41	66.25
1000.00	0.50	2005347.59	1.00	0.00	112.99	112.99	4.73	10000.00	5.00	4.08	1.41	66.67
1000.00	0.50	2005347.59	1.00	0.00	159.79	159.79	5.07	10000.00	5.00	4.08	1.41	66.67
1000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	5.00	4.08	1.41	66.67
1000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	5.00	4.08	1.41	70.00
1000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	5.00	4.08	1.41	70.00
1000.00	8.00	10695187.17	10.00	2.30	369.02	36.90	3.61	10000.00	5.00	4.08	1.41	71.25
1000.00	0.50	2005347.59	1.00	0.00	112.99	112.99	4.73	10000.00	5.00	4.08	1.41	75.00

1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	250000.00	5.00	4.08	1.41	76.25
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	250000.00	5.00	4.08	1.41	76.25
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	250000.00	5.00	4.08	1.41	76.25
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	10000.00	5.00	4.08	1.41	76.25
1000.00	0.50	668449.20	1.00	0.00	119.10	119.10	4.78	10000.00	5.00	4.08	1.41	76.25
1000.00	0.50	668449.20	1.00	0.00	65.23	65.23	4.18	10000.00	5.00	4.08	1.41	76.25
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	10000.00	5.00	4.08	1.41	76.25
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	10000.00	5.00	4.08	1.41	76.25
1000.00	0.50	668449.20	1.00	0.00	92.25	92.25	4.52	10000.00	5.00	4.08	1.41	76.25
1000.00	8.00	39572192.51	10.00	2.30	709.82	70.98	4.26	10000.00	5.00	4.08	1.41	77.00
1000.00	0.50	2005347.59	1.00	0.00	159.79	159.79	5.07	5000.00	5.00	4.08	1.41	77.75
1000.00	0.50	668449.20	1.00	0.00	130.47	130.47	4.87	25000.00	5.00	4.08	1.41	78.75
1000.00	0.50	668449.20	1.00	0.00	130.47	130.47	4.87	250000.00	5.00	4.08	1.41	78.75
1000.00	1.00	1336898.40	1.00	0.00	130.47	130.47	4.87	10000.00	5.00	4.08	1.41	78.75
1000.00	0.50	668449.20	1.00	0.00	130.47	130.47	4.87	2500.00	5.00	4.08	1.41	80.00
1000.00	1.00	2673796.79	1.00	0.00	184.51	184.51	5.22	10000.00	5.00	4.08	1.41	80.63
1000.00	4.00	2673796.79	1.00	0.00	184.51	184.51	5.22	10000.00	5.00	4.08	1.41	81.00
1000.00	2.00	2673796.79	1.00	0.00	184.51	184.51	5.22	10000.00	5.00	4.08	1.41	81.25
1000.00	2.00	2673796.79	1.00	0.00	184.51	184.51	5.22	10000.00	5.00	4.08	1.41	81.25
1000.00	0.50	2005347.59	1.00	0.00	159.79	159.79	5.07	5000.00	5.00	4.08	1.41	82.50
1000.00	4.00	5347593.58	1.00	0.00	260.94	260.94	5.56	10000.00	5.00	4.08	1.41	82.50
1000.00	8.00	10695187.17	1.00	0.00	369.02	369.02	5.91	10000.00	5.00	4.08	1.41	83.75
1000.00	8.00	39572192.51	1.00	0.00	709.82	709.82	6.57	10000.00	5.00	4.08	1.41	84.12
1000.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	5.00	4.08	1.41	85.83
1000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	50.00	5.00	1.61	1.67
1000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	25000.00	50.00	5.00	1.61	2.33
1000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	50.00	5.00	1.61	4.17
1000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	50.00	5.00	1.61	5.00
1000.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	50.00	5.00	1.61	9.17
1000.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	50.00	5.00	1.61	13.67
1000.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	50.00	5.00	1.61	16.67
1000.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	50.00	5.00	1.61	17.50
1000.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	50.00	5.00	1.61	21.67
1000.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	50.00	5.00	1.61	35.83
1000.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	50.00	5.00	1.61	43.33
1000.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	50.00	5.00	1.61	51.67
1000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	25000.00	50.00	5.00	1.61	62.67
1000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	50.00	5.00	1.61	63.33
1000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	50.00	5.00	1.61	65.83
1000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	50.00	5.00	1.61	66.67
1000.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	50.00	5.00	1.61	81.67
1500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	5.00	6.12	1.81	0.83
1500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	5.00	6.12	1.81	3.33
1500.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	5.00	6.12	1.81	8.33
1500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	5.00	6.12	1.81	8.33
1500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	5.00	6.12	1.81	15.00
1500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	5.00	6.12	1.81	15.00
1500.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	5.00	6.12	1.81	20.00
1500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	5.00	6.12	1.81	30.83
1500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	5.00	6.12	1.81	39.17
1500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	5.00	6.12	1.81	39.17
1500.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	5.00	6.12	1.81	50.00
1500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	5.00	6.12	1.81	57.50
1500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	5.00	6.12	1.81	60.83
1500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	5.00	6.12	1.81	60.83
1500.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	5.00	6.12	1.81	76.67
1500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	25.00	6.67	1.90	0.00
1500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	25.00	6.67	1.90	2.50
1500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	25.00	6.67	1.90	2.83
1500.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	25.00	6.67	1.90	7.67
1500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	25.00	6.67	1.90	7.67
1500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	25.00	6.67	1.90	14.17
1500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	25.00	6.67	1.90	14.17
1500.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	25.00	6.67	1.90	18.33
1500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	25.00	6.67	1.90	28.67
1500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	25.00	6.67	1.90	37.50
1500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	25.00	6.67	1.90	37.50
1500.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	25.00	6.67	1.90	48.33
1500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	25.00	6.67	1.90	55.33
1500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	25.00	6.67	1.90	59.16
1500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	25.00	6.67	1.90	59.17
2000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	5.00	8.16	2.10	0.00
2000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	5.00	8.16	2.10	2.50
2000.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	5.00	8.16	2.10	5.00

2000.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	5.00	8.16	2.10	5.33
2000.00	4.00	2673796.79	50.00	3.91	184.51	3.69	1.31	10000.00	5.00	8.16	2.10	6.00
2000.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	5.00	8.16	2.10	11.67
2000.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	5.00	8.16	2.10	16.67
2000.00	4.00	2673796.79	25.50	3.24	184.51	7.24	1.98	10000.00	5.00	8.16	2.10	22.50
2000.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	5.00	8.16	2.10	25.00
2000.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	5.00	8.16	2.10	35.00
2000.00	4.00	2673796.79	10.00	2.30	184.51	18.45	2.92	10000.00	5.00	8.16	2.10	45.00
2000.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	5.00	8.16	2.10	45.00
2000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	5.00	8.16	2.10	52.50
2000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	5.00	8.16	2.10	55.83
2000.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	5.00	8.16	2.10	70.00
2000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	25.00	8.89	2.18	0.00
2000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	25.00	8.89	2.18	1.67
2000.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	25.00	8.89	2.18	4.33
2000.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	25.00	8.89	2.18	5.17
2000.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	25.00	8.89	2.18	10.83
2000.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	25.00	8.89	2.18	16.33
2000.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	25.00	8.89	2.18	24.17
2000.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	25.00	8.89	2.18	33.00
2000.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	25.00	8.89	2.18	43.33
2000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	25.00	8.89	2.18	51.67
2000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	25.00	8.89	2.18	54.17
2000.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	25.00	8.89	2.18	69.16
500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	200.00	10.00	2.30	0.00
1000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	150.00	10.00	2.30	0.00
1000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	150.00	10.00	2.30	0.00
1500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	100.00	10.00	2.30	0.00
1500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	100.00	10.00	2.30	0.00
2000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	50.00	10.00	2.30	0.00
500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	25000.00	200.00	10.00	2.30	0.83
1000.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	25000.00	150.00	10.00	2.30	0.83
2000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	50.00	10.00	2.30	0.83
1000.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	150.00	10.00	2.30	1.67
1500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	100.00	10.00	2.30	1.67
500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	200.00	10.00	2.30	2.50
500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	10000.00	200.00	10.00	2.30	2.50
1000.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	150.00	10.00	2.30	3.33
1000.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	25000.00	150.00	10.00	2.30	3.33
1000.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	150.00	10.00	2.30	3.33
1500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	100.00	10.00	2.30	3.33
2000.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	50.00	10.00	2.30	3.33
2000.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	50.00	10.00	2.30	4.00
1500.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	100.00	10.00	2.30	4.17
500.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	200.00	10.00	2.30	5.83
1000.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	150.00	10.00	2.30	7.50
1000.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	150.00	10.00	2.30	7.50
500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	25000.00	200.00	10.00	2.30	8.33
1500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	100.00	10.00	2.30	9.17
1500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	100.00	10.00	2.30	9.17
2000.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	50.00	10.00	2.30	10.00
1500.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	100.00	10.00	2.30	11.67
500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	200.00	10.00	2.30	11.67
500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	200.00	10.00	2.30	13.33
500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	10000.00	200.00	10.00	2.30	13.33
2000.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	50.00	10.00	2.30	13.83
500.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	200.00	10.00	2.30	16.67
1000.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	25000.00	150.00	10.00	2.30	22.50
1500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	100.00	10.00	2.30	22.50
2000.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	50.00	10.00	2.30	22.50
1000.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	150.00	10.00	2.30	23.33
500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	200.00	10.00	2.30	26.67
500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	25000.00	200.00	10.00	2.30	30.00
1000.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	150.00	10.00	2.30	30.00
1000.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	150.00	10.00	2.30	30.00
1500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	100.00	10.00	2.30	30.83
1500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	100.00	10.00	2.30	30.83
2000.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	50.00	10.00	2.30	31.67
1000.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	150.00	10.00	2.30	36.33
500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	200.00	10.00	2.30	37.50
500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	10000.00	200.00	10.00	2.30	37.50
1500.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	100.00	10.00	2.30	40.00
500.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	200.00	10.00	2.30	41.00
2000.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	50.00	10.00	2.30	41.67

2000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	50.00	10.00	2.30	49.33
1000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	25000.00	150.00	10.00	2.30	50.83
1500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	100.00	10.00	2.30	50.83
1000.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	150.00	10.00	2.30	51.67
2000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	50.00	10.00	2.30	52.33
1500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	100.00	10.00	2.30	53.33
1500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	100.00	10.00	2.30	53.33
1000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	150.00	10.00	2.30	55.00
1000.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	150.00	10.00	2.30	55.00
500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	25000.00	200.00	10.00	2.30	56.67
500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	200.00	10.00	2.30	57.50
500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	200.00	10.00	2.30	60.83
500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	10000.00	200.00	10.00	2.30	61.67
2000.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	50.00	10.00	2.30	66.67
1500.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	100.00	10.00	2.30	68.33
1000.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	150.00	10.00	2.30	69.17
500.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	200.00	10.00	2.30	75.83
2500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	5.00	10.20	2.32	0.00
2500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	5.00	10.20	2.32	0.83
2500.00	0.50	2005347.59	25.50	3.24	41312.75	1620.11	7.39	10000.00	5.00	10.20	2.32	3.33
2500.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	5.00	10.20	2.32	4.17
2500.00	2.00	8021390.37	25.50	3.24	82625.50	3240.22	8.08	25000.00	5.00	10.20	2.32	10.00
2500.00	5.00	20053475.94	25.50	3.24	130642.38	5123.23	8.54	25000.00	5.00	10.20	2.32	13.67
2500.00	0.50	2005347.59	10.00	2.30	41312.75	4131.27	8.33	10000.00	5.00	10.20	2.32	21.67
2500.00	2.00	8021390.37	10.00	2.30	82625.50	8262.55	9.02	25000.00	5.00	10.20	2.32	31.67
2500.00	5.00	20053475.94	10.00	2.30	130642.38	13064.24	9.48	25000.00	5.00	10.20	2.32	42.33
2500.00	0.50	2005347.59	1.00	0.00	41312.75	41312.75	10.63	10000.00	5.00	10.20	2.32	49.17
2500.00	2.00	8021390.37	1.00	0.00	82625.50	82625.50	11.32	25000.00	5.00	10.20	2.32	52.50
2500.00	5.00	20053475.94	1.00	0.00	130642.38	130642.38	11.78	25000.00	5.00	10.20	2.32	66.67
2500.00	2.00	8021390.37	50.00	3.91	82625.50	1652.51	7.41	25000.00	25.00	11.11	2.41	0.00
2500.00	0.50	2005347.59	50.00	3.91	41312.75	826.25	6.72	10000.00	25.00	11.11	2.41	0.00
2500.00	5.00	20053475.94	50.00	3.91	130642.38	2612.85	7.87	25000.00	25.00	11.11	2.41	2.67

Figure B-1. Partial compilation of model results.

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BIOGRAPHICAL SKETCH

Christopher J. Brown is a senior civil engineer working for the U.S. Army Corps of Engineers in Jacksonville, Florida. During his 15-year tenure with the Corps of Engineers, Mr. Brown has worked on a variety of water resources and hazardous waste remediation projects including the Delaware River Main Channel Deepening Study, the Little Mill Creek Flood Control Study, Prompton Dam Modification, Lipari Landfill Superfund Site, B.R.O.S. Superfund Site, Modified Water Deliveries to Everglades National Park, and the Comprehensive Everglades Restoration Project.

Mr. Brown earned a B.S. in civil engineering from Temple University in Philadelphia, a M.S. in civil engineering from Villanova University in Villanova, PA, and is a candidate for Ph.D. in civil engineering from the University of Florida in Gainesville, FL. Mr. Brown's research focus at University of Florida has been the improvement of planning methodology for aquifer, storage and recovery (ASR) projects around the globe.

Mr. Brown is married and resides in Fruit Cove, Florida, located south of Jacksonville. He enjoys spending time with his wife, LJ, and numerous outdoor activities including biking, hiking, running, and swimming. Mr. Brown has also traveled across the United States visiting over 35 national parks and monuments. After graduation from the University of Florida, Mr. Brown hopes to continue working on the Everglades restoration as well as perform occasional consulting work on ASR projects located around the world.