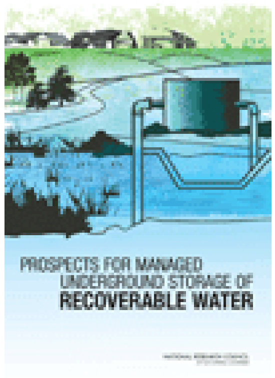

Prospects for Managed Underground Storage of Recoverable Water

Committee on Sustainable Underground Storage of Recoverable Water, National Research Council

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PROSPECTS FOR MANAGED UNDERGROUND STORAGE OF **RECOVERABLE WATER**

Committee on Sustainable Underground Storage of Recoverable Water

Water Science and Technology Board

Division on Earth and Life Studies

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Preface

As I write this, news articles from the last two weeks include "Shrinking Reservoirs Raise Concerns for Water storage" (*Hutchinson* [Kansas] *News*, July 2, 2007), "Solution to Our Dwindling Water Supply Lies Below: Subsurface Water Storage is an Economic Way to Address Seasonal Shortages" (*Seattle Daily Journal of Commerce*, June 28, 2007), and "Naples' Plan for Water Storage Well Hits Snag" (*Naples* [Florida] *News*, June 20, 2007). Virtually every day's newspaper articles describe difficult choices that have to be made in water management all over the country and the possible role of underground storage of water in addressing these challenges.

Putting away water in times of abundance and retrieving it in times of need is nothing new. Traditionally water has been stored in surface reservoirs. However, problems associated with surface reservoirs, such as, evaporative losses, sediment accumulation, land consumption, and ecological impacts, have driven water managers to seek alternatives for providing reliable water supplies. One of these alternatives is storing water underground. The number of these projects has grown in the last two decades. From 3 underground storage systems in 1983, the number jumped to 72 in late 2005 with projections indicating continued increases. Many of these projects are being developed in areas where water supply crises are anticipated in the future. Throughout the United States, freshwater supplies may be hard pressed to meet projected needs for a variety of reasons, such as overdrafted aquifers, increased competitive use of water, and climate change. While there is no indication of any slowdown in the number of projects being planned and developed, many scientific, operational, and institutional issues remain to be addressed–hence, the timing of this study.

This project traces its roots to a strategic planning session of the Water Science and Technology Board (WSTB) of the National Research Council (NRC), which rated the topic among its highest priorities. In 2003, along with the AWWA Research Foundation (AwwaRF), the WSTB organized a planning workshop that brought together more than two dozen scientists and engineers to evaluate the potential for underground storage to contribute clean and reliable water. The planning workshop also helped to highlight priority issues that are reflected in the study's statement of task (see Summary Box S-1).

Augmenting freshwater supply by underground storage is such a pressing concern that when the WSTB sought support for the study, a wide range of sponsorship was generated from federal, state, and private organizations. Sponsors for this study reflect the wide interests in the potential for managing underground storage. We would like to thank the following for supporting the study and providing staff, assistance, data, and information in a timely and helpful manner to the committee: AwwaRF, WateReuse Foundation, U.S. Geological Survey (USGS), The CALFED Bay-Delta Program and the California Department of Water Resources Conjunctive Water Management Branch, the City of Phoenix, the Inland Empire Utilities Agency, the Sanitation Districts of Los Angeles, the Chino Basin Watermaster, the Water Replenishment District of Southern California, the National Science Foundation, and the NRC President's Committee of the National Academies.

In developing this report, the committee received advice and input from Richard Atwater, Inland Empire Utilities Agency; Robert Hultquist, California Department of Health Services; John Izbicki, USGS; Paul Kinshella, Phoenix Water District; Jeff Mosher, WateReuse Foundation; Hoover Ng, Water Replenishment District of Southern California; Chris Pitre, Golder Associates; David Pyne, ASR Systems; Steve Ragone, National Ground Water Association; Judy Richtar, Florida Department of Environmental Protection; Martha Rincon, Los Angeles County Sanitation District; Shane Snyder, Southern Nevada Water Authority; John Taylor, Environmental Protection Agency (EPA) Region V; Ryan Ulrich, AwwaRF; Mark Wildermuth, Chino Basin Watermaster; Greg Woodside, Orange County Water District; and Gary Zeigler, consultant. We also thank all those who took the time to share their perspectives and expertise by participating in meetings, and field trips and by sending their written comments.

The accomplishment of this report depended upon highly devoted staff and the efforts of the committee members. I thank Will Logan and Ellen de Guzman, the NRC study director and research associate, respectively, for their input to this project. Ellen and Will planned the committee meetings, compiled information, interacted with the committee members to maximize their contributions and writings, offered insightful comments and directions, and synthesized and edited the final report. I thank the committee members who took the time to share their perspectives and knowledge about underground storage systems and their experiences with water management. It is rewarding to work with such a talented and articulate group of professionals.

This report has been reviewed in draft form by individuals chosen for their diverse perspectives and technical expertise, in accordance with procedures approved by the NRC's Report Review Committee. The purpose of this independent review is to provide candid and critical comments that will assist the institution in making its published report as sound as possible and to ensure that the report meets institutional standards for objectivity, evidence, and responsiveness to the study charge. The review comments and draft manuscript remain confidential to protect the integrity of the deliberative process. We wish to thank the following individuals for their review of this report: ; Jean Bahr, University of Wisconsin, Madison; Michael Brinkmann, San Antonio Water System, Texas; Christopher Brown, Golder Associates, Jacksonville, Florida; Peter Dillon, CSIRO, Center for Groundwater Studies, Australia; Charles Haas, Drexel University, Philadelphia, Pennsylvania; Tanya Heikkila, Columbia University, New York, N.Y.; William Jury, University of California, Riverside; Ronald Lacewell, Texas A&M University, College Station; R. David Pyne, ASR Systems LLC, Gainesville, Florida; John Taylor, U.S. EPA, Chicago, Illinois, and; Madeline Schreiber, Virginia Polytechnic Institute and State University, Blacksburg.

Although the reviewers listed above have provided many constructive comments and suggestions, they were not asked to endorse the conclusions and recommendations nor did they see the final draft of the report before its release. The review of this report was overseen by Jerome Gilbert, J. Gilbert, Inc. Appointed by the National Research Council, he was responsible for making certain that an independent examination of this report was carried out in accordance with institutional procedures and that all review comments were carefully considered. Responsibility for the final content of this report rests entirely with the authoring committee and the institution.

Edward J. Bouwer
Committee Chair

Contents

Summary

Pressure on freshwater supplies will increase to meet anticipated needs for municipal and industrial uses, agricultural irrigation, and environment protection in the coming years. Certain conditions such as increasing population, changing land use, reallocation of existing water resources, reduction of snowpack, and overdrafting of aquifers will require tapping into other non-traditional sources of water. While other water management strategies have been used to increase freshwater supply through importation or desalination, improving water efficiency through technology and conservation, and reuse of treated wastewater, the potential for managed underground systems to sustain future water supplies is considerable.

With or without the other strategies, there is already a need for temporary detention and storage of water during times of abundance and recovery that water in times of scarcity. The traditional practice of storing water aboveground has been met with several challenges such as evaporative losses, sediment accumulation, land consumption, high cost, and ecological impact. Because of these factors there is increasing interest in storing recoverable water underground as part of a larger water management strategy. This has brought with it, however, its own set of challenges, such as costs to design, construct, and monitor the system; loss of some percentage of the water; chemical reactions with aquifer materials; ownership issues; and environmental impacts.

The source water for underground storage may come from streams or groundwater, water reclamation plants, or other sources and be recharged through different methods. After recovery, it may be used for potable, industrial, agricultural, environmental, and other purposes. For this report, the term managed underground storage (MUS) is used to refer to this purposeful recharge of water into an aquifer system for intended recovery and use as component of long-term water resource management.

The growing importance of the topic emphasized the need to study the state of the knowledge and identify the research and education needs and priorities for Managed Underground Storage of Recoverable Water. In 2003, the National Research Council (NRC) along with the AWWA Research Foundation (AwwaRF) organized a planning meeting for a consensus study on the topic. The feedback received during this planning meeting was instrumental in formulating a statement of task (Box S-1) for a follow-on study, whose results are summarized in this report.

In early 2005, the authoring committee for this report met for the first time to identify research and education needs and priorities in underground storage technology and implementation. Members represented multidisciplinary

BOX S-1
Statement of Task

The proposed study will provide an overview of some of the research and education needs and priorities concerning sustainable underground storage[1] technology and implementation. It will also assess geological, geochemical, biological, engineering, and institutional factors that may affect the performance of such projects, based in part on a review and evaluation of existing projects.

Specifically, the study will assess and make recommendations with respect to research and education needs on the following questions:

- What research needs to be done to develop predictors of performance for underground storage projects based on the character of the recharge water in terms of contaminants, disinfectants, and microbes, the hydrogeology and major ion geochemistry of the source water and the aquifer, and the well or basin characteristics?
- What are the long-term impacts of underground storage on aquifer use—hydraulic, geotechnical, geochemical, adsorptive capacity of contaminants—at wellhead and regional scales, and can these impacts be ameliorated?
- What physical, chemical, and geological factors associated with underground storage of water may increase or decrease human and environmental health risks concerning microbes, inorganic contaminants such as nitrite, disinfectant by-products, endocrine disruptors, personal care products, pharmaceuticals, and other trace organic compounds?
- Are there any chemical markers or surrogates that can be used to help assure regulators and the public of the safety of water for groundwater recharge? What should we monitor and at what spatial and temporal scales?
- What are the challenges and potential for incorporating sustainable underground storage projects into current systems approaches to water management for solving public and environmental water needs?
- How do the institutional, regulatory and legal environments at federal, state, and local levels encourage or discourage sustainable underground storage?

expertise in groundwater and surface water hydrology, inorganic and organic hydrogeochemistry and biogeochemistry, risk assessment, environmental and water resources engineering, water reuse, and natural resource economics and law.

The potentially widespread implication of the study is apparent in its sponsors, which represent water utilities, water associations, federal and state agencies, and science organizations: the American Water Works Association Research Foundation, the WateReuse Foundation, the U.S. Geological Survey (USGS), the CALFED Bay-Delta Program and the California Department of Water Resources Conjunctive Water Management Branch, the City of Phoenix, the Inland Empire Utilities Agency, the Sanitation Districts of Los Angeles County, the Chino Basin Watermaster, and the NRC President's Committee of the National Academies.

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[1] In this report the term “managed underground storage” is used instead of “sustainable underground storage.”

An overall evaluation and a summary of the key conclusions and recommendations of the study follow.

OVERALL EVALUATION

Conclusion: The challenges to sustaining present and future water supplies are great and growing. The present overdrafting of aquifers and overallocation of rivers in many regions is a clear indication of these challenges, but the former also creates in many cases the underground storage potential needed to accommodate MUS systems. Thus, demand for water management tools such as MUS is likely to continue to grow (Chapter 1).

Conclusion: Some simple forms of MUS have been used for millennia, and even the most recent development–aquifer storage and recovery–now has about four decades of history behind it. These systems use water from a variety of sources such as surface water, groundwater, treated effluent, and occasionally stormwater. They recharge groundwater through recharge basins, vadose zone wells, and direct recharge wells. The water is stored in a wide spectrum of confined and unconfined aquifer types, from unconsolidated alluvial deposits to limestones and fractured volcanic rocks. Recovery typically is achieved through either extraction wells or dual-purpose recharge and recovery wells, but occasionally is achieved via natural discharge of the water to surface waterbodies. Finally, the recovered water is used for drinking water, irrigation, industrial cooling, and environmental and other purposes. There is, therefore, adequate experience from which to draw some general conclusions about the degree to which MUS systems are successful in meeting their stated goals and the challenges and difficulties that some of them face (Chapter 2).

Conclusion: Although failures have occurred and the potential for contaminating groundwater is a considerable risk, most MUS systems have successfully achieved their stated purposes. In fact, there are MUS systems that have functioned without major problems for decades. However, increasing efforts to use karst and fractured aquifers for storage will increase the potential for failures. Chemical reactivity of the aquifer in the former case and uncertainty over flow paths in either case are much greater and the treatment potential is lower compared to alluvial aquifers. Learning from past positive and negative performance will help guide development of the many new MUS systems that are under consideration (Chapter 7).

Recommendation: Given the growing complexity of the nation's water management challenges, and the generally successful track record of managed underground storage in a variety of forms and environments, MUS should be seriously considered as a tool in a water manager's arsenal (Chapters 1-7).

Conclusion: In the future, multiple strategies are likely to be needed to manage water supplies and meet demands for water in the face of scarcity.

Various water conservation and management strategies, including transfers and water recycling, can be used to stretch available water supplies. However, each of these has its rate of delivery limits. Water storage facilities will continue to be an essential component of water management, particularly in areas where water availability varies greatly over seasons or years, such as the arid Southwest. Integrated strategies will be needed in which all measures for improving water quality and managing water scarcity are considered and, if appropriate, employed in a balanced, systematic fashion. Seasonal to multi-year storage of water will often be a necessary component of such strategies.

Recommendation: In anticipating, planning for, and developing MUS projects, water managers should consider them in a watershed and regionally based context and as part of the overall water management strategies (Chapter 7).

HYDROGEOLOGICAL ISSUES

Conclusion: To facilitate the siting and implementation of MUS systems, maps of favorable aquifers and hydrogeological characteristics can be prepared using three-dimensional (3-D) capable geographical information systems (GIS). At a regional or statewide scale, such GIS maps can help visualize and characterize major aquifers for future development of MUS systems, map and analyze regional changes in head and flow patterns, and facilitate comprehensive, regional water resources management. At a project scale, they can aid in establishing the design, spacing, orientation, and capacity of wells and recharge basins, evaluating their impact on the environment and existing users, estimating the critical pressure for rock fracturing, visualizing the movement of stored water throughout the system (especially useful for systems with waters of varying density or quality), and evaluating the extent of potential water quality changes in the aquifer during storage and movement.

Recommendation: States, counties, and water authorities considering MUS should consider incorporating 3-D capable geographical information systems along with existing hydrogeologic, geochemical, cadastral, and other data in (1) regional mapping efforts to identify areas that are, or are not, likely to be favorable for development of various kinds of MUS systems, and (2) project conception, design, pilot testing, and adaptive management (Chapter 3).

Conclusion: Long-term local and regional impacts of MUS systems on both native groundwater and surface water have been recognized, including changes in groundwater recharge, flow, and discharge, and effects on aquifer matrix such as compaction of confining layers or clay interlayers during recharge and recovery cycles.

Recommendation: Monitoring and modeling should be performed to predict likely effects–positive or negative–of MUS systems on the physical system, including inflows, storage, and outflows. Appropriate measures can and should be taken to minimize negative effects during operations (Chapter 3).

Conclusion: Groundwater numerical modeling at regional and/or high-resolution local scales provides a cost-effective tool for planning, design, and operation of a MUS system.

Recommendation: Analyses using groundwater flow and solute transport modeling should become a routine part of planning for, designing, and adaptively operating MUS systems. Uncertainty analysis should also be incorporated into prediction of a system's short- and long-term performance, especially regarding the expected values of recovery efficiency and storage capacity (Chapter 3 and 4).

Specific Research Recommendations: In addition to the topics above, research is particularly needed, and should be conducted, in the following areas (Chapter 3):

- *Hydrologic feasibility.* This includes (1) a lack of knowledge about storage zones and areas favorable for recharge for major aquifers in the United States; (2) limited understanding of how aquifer heterogeneity, scale effects, and other physical, chemical, and biological properties impact recharge rate and recovery efficiency of the MUS system; (3) a lack of understanding of matrix behavior, especially fractured aquifers, during injection vs. withdrawal tests (e.g., expansion vs. compaction) to prevent or limit artificially induced deformation of the aquifer matrix; (4) a need to develop tools to analyze non-Darcian flow around recharge wells to avoid poor design of recharge wells; and (5) need for overall characterization, system recovery efficiency, optimum placement of monitoring wells, recharge and pumping impacts, and hydraulic fracturing in an aquifer with dual porosity.
- *Impacts of MUS systems on surface water*. How, in terms of both quantity and timing, might a surface spreading or well recharge facility affect the flow of neighboring streams? What would be the hydrologic, ecological, and legal consequences of this interaction between the MUS system and surface water? An integrated or system approach should be developed and employed for assessing such impacts.
- *Technology enhancement and methodology development for determining hydrological properties of the aquifers and their impacts on performance of the MUS system.* These include (1) surface and borehole geophysical methods to determine hydrological properties and the extent of recharge water volumes during cycle testing, (2) optimization of cycle test design (frequency, duration, and intensity) to improve performance of MUS systems for various hydrological settings, (3) better conceptual models for delineation of storage zone and recovery zone, and (4) better understanding of non-Darcian flow near wells through experimental study and field monitoring, and further development of

theories and numerical models to assess the interaction of stored water (especially urban runoff) with native groundwater.

WATER QUALITY CONSIDERATIONS

Conclusion: There is a substantial body of work documenting improvements in water quality that can occur in an MUS system, particularly those that involve surface spreading. The subsurface has, to a greater or lesser extent, the capacity to attenuate many chemical constituents and pathogens via physical (e.g., filtration and sorption), chemical, and biological processes. In places where the groundwater quality is saline or otherwise poor, the implementation of MUS will likely improve overall groundwater quality and provide a benefit to the aquifer.

However, the type of source water used for recharge along with subsurface properties and conditions influences the extent of treatment and the effects on native groundwater quality. Therefore, a thorough knowledge of the source water chemistry and mineralogy of the aquifer is requisite to embarking on any MUS project. It is important to establish whether the mixing of source water and native groundwater, as well as chemical interaction with aquifer materials, yields compatible and acceptable effects on water quality.

Recommendation: A thorough program of aquifer and source water sampling, combined with geochemical modeling, is needed for any MUS system to understand and predict its medium- and long-term chemical behavior and help determine the safety and reliability of the system (Chapter 4).

Conclusion: A better understanding of the contaminants that might be present in each of the potential sources of recharge water is needed, especially for underutilized sources of water for MUS, such as stormwater runoff from residential areas. Limited data exist on the use of urban stormwater for MUS systems. Consistent with an earlier NRC report (1994), urban stormwater quality is highly variable and caution is needed in determining that the water is of acceptable quality for recharge.

Recommendation: Research should be conducted to evaluate the variability of chemical and microbial constituents in urban stormwater and their behavior during infiltration and subsurface storage to establish the suitability of combining MUS with stormwater runoff (Chapter 4).

Conclusion: The presence and behavior of “emerging” contaminants (e.g., endocrine disrupting compounds, pharmaceuticals, and personal care products) is of concern, especially with reclaimed wastewater. However, the concern about these compounds is not unique to MUS systems. Surface waters and groundwaters around the nation carry the same kind of chemicals, and surface water treatment systems are not normally designed to address them.

Recommendation: Basic and applied research on emerging contaminants that has begun at a national scale should be encouraged, and MUS programs will be among the many beneficiaries of such investigations (Chapter 4).

Conclusion: A better understanding is needed of potential removal processes for microbes and contaminants in the different types of aquifer systems being considered for MUS. These studies are necessary to assess spatial and temporal behavior during operation of an MUS system. This research could reduce uncertainty regarding the extent of chemical and microbial removal in MUS systems. In addition, this information could help reduce impediments to public acceptance of a wide variety of source waters for MUS.

Conclusion: In particular, changes in reduction-oxidation (redox) conditions in the subsurface are common and often important outcomes of MUS operation. These changes can have both positive and negative influences on the physical properties and the chemical and biological reactivity of aquifer materials. For example, the existence of both oxidizing and reducing conditions might enhance the biodegradation of a suite of trace organic compounds of concern or, conversely, lead to accumulation of an intermediate product of concern. Redox changes can cause dissolution-precipitation or sorption-desorption reactions that lead to adverse impacts on water quality or clogging of the aquifer; however, such precipitation reactions can also sequester dissolved contaminants.

Recommendation: Additional research should be conducted to understand potential removal processes for various contaminants and microbes and, particularly, to determine how changes in redox conditions influence the movement and reactions for many inorganic and organic constituents. Specific areas of research that are recommended include (1) bench-scale and pilot studies along with geochemical modeling to address potential changes in water quality with variable physical water conditions (pH, oxidation potential [Eh], and dissolved oxygen [DO]); and (2) examination of the influence of sequential aerobic and anaerobic conditions or alternating oxidizing and reducing conditions on the behavior of trace organic compounds in MUS systems, especially during storage zone conditioning (Chapter 4).

Conclusion: Molecular biology methods have the potential for rapid identification of pathogens in water supplies. These noncultivable techniques have not been tested in a meaningful way to address background and significance of the findings. False negatives and false positives remain an issue that needs to be addressed.

Recommendation: Research should be conducted to address the approaches and specific applicability of molecular biology methods for pathogen identification, particularly interpretation of results that cannot determine viability, for the different types of source waters and aquifer systems being considered for MUS (Chapter 4).

Conclusion: Pathogen removal or disinfection is often required prior to storing water underground. If primary disinfection is achieved via chlorination, disinfection by-products (DBPs) such as trihalomethanes and haloacetic acids are formed. These have been observed to persist in some MUS systems. However, chlorine is the most cost-effective agent for control of biofouling in recharge wells; hence, it may not be possible to eliminate entirely the use of chlorine in MUS systems (e.g., periodic pulses of chlorine to maintain injection rates).

Recommendation: To minimize formation of halogenated DBPs, alternatives to chlorination should be considered for *primary* disinfection requirements, such as ultraviolet, ozone, or membrane filtration (Chapter 4).

LEGAL, ECONOMIC, AND OTHER INSTITUTIONAL CONSIDERATIONS

Conclusion: Some states have created statutory schemes that are tailored to MUS projects, This approach is desirable because of the novel questions posed. For example, a state may find it desirable that withdrawals from an MUS project be done over a longer period than a traditional water right might provide, or that MUS be allowed despite the “junior” status of the right’s holder. States can anticipate these adjustments to traditional water rights as appropriate.

Recommendation: While a comprehensive approach has advantages, at a minimum states should define property rights in water used for recharge, aquifer storage, and withdrawn water, to provide clarity and assurance to MUS projects (Chapter 5).

Conclusion: The federal regulatory requirements for MUS are inconsistent with respect to treatment of similar projects. Federal Underground Injection Control (UIC) regulation addresses only projects that recharge or dispose of water directly to the subsurface through recharge wells, while infiltration projects are regulated by state governments whose regulatory standards may vary. The appropriateness of regulation through the UIC program has been questioned by states with active aquifer storage and recovery (ASR) regulatory programs. Also, there are inconsistencies between the Clean Water Act and the Safe Drinking Water Act that impact MUS systems. For example, some jurisdictions try to control surface water contamination problems by diverting polluted water from aboveground to groundwater systems. This approach may undermine MUS programs by putting contaminants underground without appropriate controls.

Recommendation: Federal and state regulatory programs should be examined with respect to the need for continued federal involvement in regulation, the necessity of a federal baseline for regulation, and the risks presented by inadequate state regulation. A model state code should be drafted that would assist states in developing comprehensive regulatory programs that reflect a scientific approach to risk (Chapter 5).

Conclusion: Regulations are, quite properly, being developed at the state level that will require a certain residence time, travel time, or travel distance for recharge water prior to withdrawal for subsequent use. However, regulations based on attenuation of a single constituent or aquifer type, such as pathogen attenuation in a homogeneous sand aquifer, may not be appropriate for a system concerned with trace organics and metals in a fractured limestone, and vice versa. Such regulations are particularly pertinent for MUS with reclaimed water.

Recommendation: Science-based criteria for residence time, travel time, or travel distance regulations for recharge water recovery should be developed. These criteria should consider biological, chemical, and physical characteristics of an MUS system and should incorporate criteria for adequate monitoring. The regulations should allow for the effects of site-specific conditions (e.g., temperature, dissolved oxygen, pH, organic matter, mineralogy) on microbial survival time or inactivation rates and on contaminant attenuation. They should also consider the time needed to detect and respond to any water quality problems that may arise (Chapter 5).

Conclusion: MUS projects can exhibit numerous and complementary economic benefits, but they also entail costs. Some of those benefits and costs are unlikely to be incorporated in the calculations of individual water users—that is, there may be spillover costs to third parties or spillover benefits that are not given market valuations. Failure to account for all benefits and costs, including ones that may not be reflected in market prices for water, can lead to underinvestment in groundwater recharge, overconsumption of water supplies, or both.

Recommendation: An economic analysis of an MUS project should capture the multiple benefits and costs of the project. MUS projects invariably entail the achievement of multiple objectives. Third party impacts, such as the environmental consequences of utilizing source water, should be included (Chapter 5).

Conclusion: Water resources development has been characterized by substantial federal and state subsidies. As water shortages intensify, the political pressure for investment in new technologies will increase.

Recommendation: To ensure optimal investment in MUS and other technologies, subsidies should be provided only when there are values that cannot be fully reflected in the price of recovered waters. An example of such a value would be an environmental benefit that accrues to the public at large. In particular, simply lowering costs should not be the justification for providing subsidies for MUS projects (Chapter 5).

Conclusion: Antidegradation is often the stated goal of water quality policies, including policies that apply to underground storage of water. For any MUS project–including storage of potable water, stormwater, and recycled water–it is important to understand how water quality differences between native groundwater and the stored water will be viewed by regulators, who are charged with satisfying those regulatory mandates. In addition to water quality factors, a broader consideration of benefits, costs, and risks would provide a more desirable regulatory approach. Therefore, weighing water quality considerations together with water supply concerns, conservation, and public health and safety needs is an essential plan of action. Rigid antidegradation policies[2] can impede MUS projects by imposing costly pretreatment requirements and may have the practical effect of prohibiting MUS, even in circumstances where the prospects of endangering human or environmental health are remote and the benefits of water supply augmentation are considerable.

Recommendation: State laws and regulations should provide regulatory agencies with discretion to consider weighing the overall benefits of MUS while resolutely protecting groundwater quality (Chapter 5).

OPERATIONAL ISSUES

Conclusion: The development of an MUS system from project conception to a mature, well functioning system is a complex, multistage operation requiring interdisciplinary knowledge of many aspects of science, technology, and institutional issues.

Recommendation: A comprehensive decision framework should be developed to assist in moving through the many stages of project development in an organized, rational way. Professionals from many fields, including chemists, geologists, hydrologists, microbiologists, engineers, economists, planners, and other social scientists should be involved in developing this framework (Chapter 6).

Conclusion: Growing experience with MUS systems indicates that hydrogeological feasibility analysis including aquifer characterization is one of several important components in their development and implementation. The benefits of doing so include establishing the hydraulic capacity, recharge rates, residence times, and recoverable fraction of the introduced water–all of which help identify the optimum design and viability of the MUS system.

Some types of aquifers have matrix, hydrogeologic, and geochemical characteristics that are better suited to MUS systems than others. For example, the aquifer characteristics may dictate recharge, storage, and recovery methods. For an unconfined aquifer, source water can be recharged into the aquifer through recharge basins, vadose zone recharge wells, and deep recharge wells. Stored water can be recovered by production wells or ASR wells, or it can enhance baseflow to neighboring streams. For confined aquifers, however, source water can only be injected through deep recharge wells, including ASR wells. The

[2] In Chapter 5, the term "rigid antidegradation policies" refers to prohibiting any change whatsoever in groundwater quality, even when both the source water and the aquifer water meet all drinking water standards. Further discussion is found in Chapter 5.

stored water is usually recovered through ASR wells or downgradient production wells. As another example, water quality benefits are likely to be greater with alluvial systems compared to fractured or dual-porosity systems.

Recommendation: Multiple factors should be assessed and monitored during design, pilot tests, and operations, including spatial and hydrogeological characterization of storage zones; temporal variation in quality and quantity of recharged, stored, and recovered water; and factors that constrain sustainability of the MUS system, including hydrogeochemical, microbiological, and economic conditions. Uncertainty reduction is the ultimate goal (Chapters 3, 4, and 6).

Conclusion: An independent advisory panel can provide objective, third-party guidance and counsel regarding design, operation, maintenance, and monitoring strategies for an MUS project. An independent panel can increase public acceptance of and confidence in the system if such trust is warranted. It can also be a catalyst for altering a plan if changes appear to be necessary.

Recommendation: Water agencies should highly consider the creation of an independent advisory panel or equivalent at an early stage of planning for an MUS system (Chapter 6).

Conclusion: Relatively little research has been done to characterize the extent of vertical migration of fine-grained particles into the sediments beneath surface spreading facilities. Likewise, the science and technology of cleaning recharge basins is not well developed.

Recommendation: New approaches should be developed to optimize surface recharge, including assessing the extent of migration of fine-grained sediment into the subsurface, its impact on the long-term sustainability of surface recharge, and more efficient methods to clean recharge basins after clogging occurs (Chapter 6).

Conclusion: Successful MUS involves careful and thorough chemical and microbiological monitoring to document system performance and evaluate the reliability of the process. Each MUS project needs real-time monitoring of the quality of the waters being introduced into underground storage and of waters being extracted from storage for use.

Recommendation: Water quality monitoring programs should be designed on a case-by-case basis to assess water quality changes for elements, compounds, and microbes of concern, optimizing the potential to document any improvement in the quality of the source water and to collect samples representing any adverse water quality changes. A proactive monitoring plan is needed to respond to emerging contaminants and increase knowledge about potential risks (Chapters 4 and 6).

Conclusion: New surrogates or indicators of pathogen and trace organic contaminant presence are needed for a variety of water quality parameters to increase the certainty of detecting potential water quality problems through monitoring. The categorization of chemicals and microorganisms into groups with similar fate and transport properties and similar behavior in treatment steps is one approach to streamline the list of potential contaminants to be monitored. It is unclear whether we can continue to rely on total coliform and *Escherichia coli* indicator bacteria to characterize the microbial quality of water as the drinking water industry has done for decades. Such methodologies will improve the ability of MUS systems of a variety of sizes to engage in sound monitoring practices.

Recommendation: Research should be conducted to understand whether we can rely on monitoring surrogate or indicator parameters as a substitute for analysis of long lists of chemicals and microorganisms (Chapter 6).

Conclusion: Surface spreading facilities sometimes require large amounts of land, particularly where large amounts of water are recharged or the geology is not ideal. Recharge well systems require less land but may have as many different factors to consider in their placement. Optimization of recharge facility placement is important but not always well understood.

Recommendation: If there is some degree of freedom in site selection for recharge wells or basins, a location suitability assessment may be useful in site optimization. Factors such as ecological suitability, existing uses of the aquifer, groundwater quality, aquifer transmissivity, road density, land use and ownership, and access to power lines can be weighed in such an analysis (Chapter 6).

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1
Introduction

Between 1990 and 2000, over 700,000 people found their way to the Las Vegas metropolitan area (http://www.census.gov/prod/2001pubs/c2kbr01-2.pdf). They found homes, settled in, and turned on their taps. Miraculously, water came out.

One day it might not. With thousands of newcomers per month moving into the area–a phenomenon repeated in other states such as Arizona and Texas–water managers are challenged as never before to do more with less. Snowpack in the western and northeastern United States appears to be decreasing (Mote et al., 2003; Hodgkins and Dudley, 2006), and groundwater overdrafting throughout the nation continues unabated in many locations. Portions of aquifers in every state along the Atlantic Coastal Plain, from New Jersey to Georgia, have had to be protected and managed to prevent continued reductions in groundwater levels, land subsidence, and saltwater intrusion.

The increasing pressures on water in the western United States have been highlighted in the U.S. Bureau of Reclamation's Water 2025 initiative (Figure 1-1). Eastern states have also moved toward planning programs to address demands related to scarce water resources due to periodic droughts, increasing populations, changing land use, and the links between water use and environmental protection (Virginia Department of Environmental Quality, 2001). Average temperatures in many regions of the country are rising and are projected to continue to do so; in such areas, both supply and evaporative losses may be headed in unhelpful directions. Conservation is an important water management tool, but a 10 percent savings of water—a significant figure—would take care of only 18 months of population growth for a city that is growing at a rate of 7 percent per year, as is the case for Las Vegas. Then what?

Historically, the answer has been to build a dam. Throughout the last few centuries, about 76,000 dams more than 2 m high were constructed on our rivers and streams (*http://crunch.tec.army.mil/nid/webpages/nid.cfm*), and many of these had seasonal or interannual water storage as their primary function. Yet with evaporation rates of 120-200 cm in states such as Arizona (see *http://www.water.az.gov/dwr*), the limited availability of land for construction, and the high environmental costs to stream and riparian wildlife, the building of dams and reservoirs scarcely seems to be an approach that will provide much relief in the future.

All of these considerations portend increasing stresses on our water supply in the coming years and increasing burdens on our water managers. New strategies for water management—with respect to both quality and quantity—will be required on a broad geographic scale. Options for addressing these issues

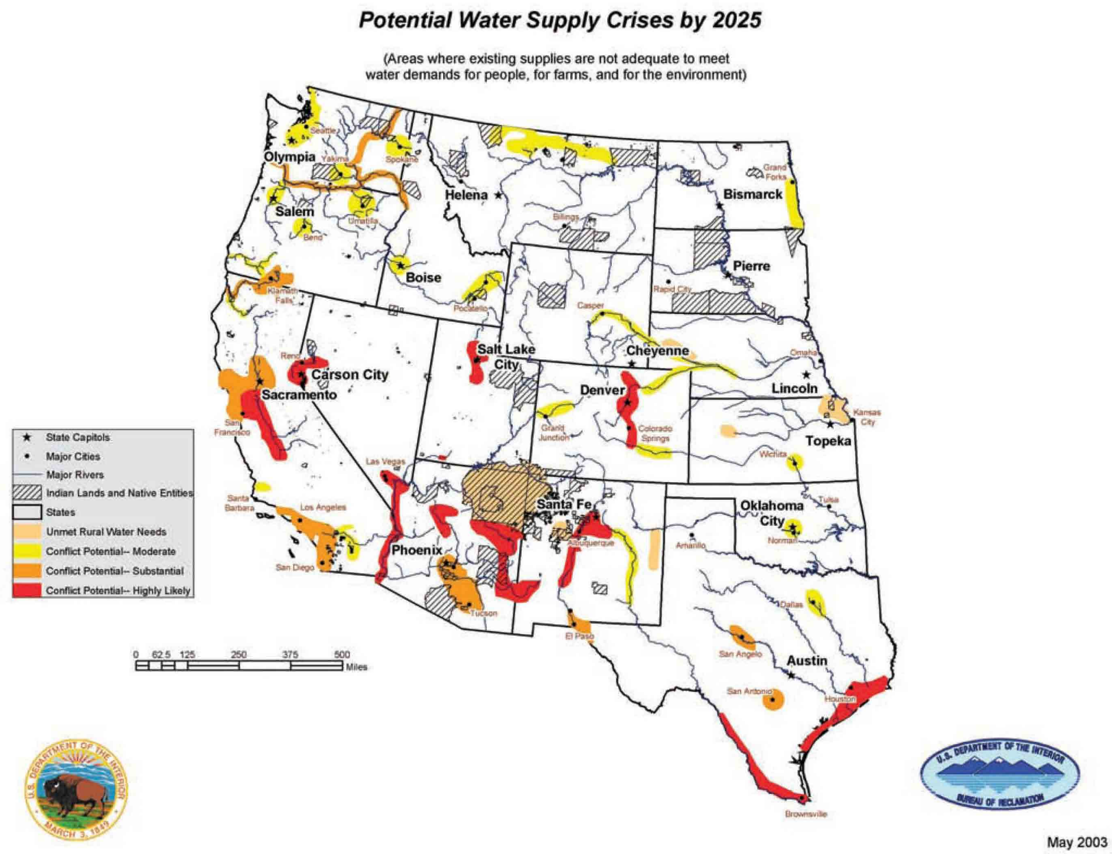

FIGURE 1-1 Potential water supply crises and conflicts in the western United States by the year 2025. SOURCE: U.S. Bureau of Reclamation. Available online at *http://www.doi.gov/ water2025/report.pdf*. Accessed September 24, 2007.

include improving water use efficiency through technology and conservation, increasing supply through importation and desalination, and reuse of treated wastewater.

With or without these strategies, however, there is often a need for temporary detention and storage of water during times of abundance for release during times of need. Because of the shortcomings often associated with storage in aboveground reservoirs–including evaporative losses, land consumption, and ecological impacts–there is increased interest in storing recoverable water underground as part of an overall water management plan. Storing surface water underground seems counterintuitive to many people, who consider this a "waste" because the water may move away from the recharge area and not raise the water table at all. The counterarguments to this are hydrogeological (see Chapter 3) and are not described here. Suffice it to say that while some recharged water may, indeed, never be recovered, the same is true for surface water stored in reservoirs. The circumstances under which groundwater storage may or may not be desirable relative to surface storage are among the primary themes of this report.

The water to be stored may come from streams or groundwater (with or without treatment at water treatment plants), water reclamation plants, stormwater, or other sources. It may be recharged through wells or infiltration basins into sands and gravels, limestones, granites, or volcanic rocks. The water may be stored for days, months, seasons, or several years. The stored water may be recovered from the aquifer by the same well that recharged it or by a downgradient well. After recovery, it may be used for drinking water, industrial purposes, golf course or lawn irrigation, agriculture, or aquatic habitat restoration.

While several terms have developed over the years to describe various aspects of this concept, with examples provided in Box 1-1, none of the existing words or expressions in the field of water management quite describes this concept in its entirety. For the purposes of capturing the full range of approaches considered in this study, the committee proposes the term “managed underground storage of recoverable water” (MUS), the rationale for which is described in Box 1-2. In this report, MUS is used to denote *purposeful recharge of water into an aquifer system for intended recovery and use as an element of long-term water resource management.*

Managed underground storage (MUS) systems would encompass both systems in which water is recharged directly using wells (including dual-purpose recharge and recovery wells) and systems that use infiltration basins. However, the term as defined would exclude riverbank filtration systems (no storage) and underground disposal of brines or recharge of water for the sole purpose of mitigating land subsidence or aquifer depletion or to prevent saltwater intrusion (no planned recovery of the water).

It is recognized, of course, that there are gray areas, such as water recharged primarily to prevent saltwater intrusion that is partially recovered on the landward side of the subsurface “mound.” Such are the hazards of creating new jargon.

BOX 1-1
Terms Used to Describe Related Water Management Approaches Involving Recharge

- ***Aquifer storage and recovery (ASR)—***injection of water into a well for storage and recovery from the same well.
- ***Aquifer storage transfer and recovery (ASTR) —***injection of water into a well for storage and recovery from a different well, generally to provide additional water treatment.
- ***Artificial recharge (AR) —***intentional banking and treatment of water in aquifers.
- ***Artificial recharge and recovery (ARR) —***recharge to and recovery of water from an aquifer; that is, both artificial recharge of the aquifer and recovery of the water for subsequent use.
- ***Augmentation pond—***water body designed to supply water to river systems at defined rates during particular times.
- ***Bank filtration—***extraction of groundwater from a well or caisson near or under a river or lake to induce infiltration from the surface water body, thereby improving and making more consistent the quality of water recovered.
- ***Conjunctive use—***combining the use of both surface and groundwater to minimize the undesirable physical, environmental, and economic effects of each solution.
- ***Dry well—***synonymous with *vadose zone well*.
- ***Infiltration basin—***synonymous with *recharge basin*.
- ***Managed (or management of) aquifer recharge (MAR)—***intentional banking and treatment of water in aquifers (synonymous with AR). MUS may be considered a subset of MAR.
- ***Recharge basin (or pond)—***a surface facility, often a large pond, used to increase the infiltration of surface water into a groundwater basin; basins require the presence of permeable soils or sediments at or near the land surface and an unconfined aquifer beneath. ***Recharge well—***a well used to directly recharge water to either a confined or an unconfined aquifer.
- ***Soil aquifer treatment (SAT)—***treated sewage effluent, known as reclaimed water, is intermittently infiltrated through infiltration ponds to facilitate nutrient and pathogen removal in passage through the unsaturated zone for recovery by wells after residence in the aquifer.
- ***Surface spreading—***recharging water at the surface through recharge basins, ponds, pits, trenches, constructed wetlands, or other systems.
- ***Spreading basin—***synonymous with *recharge basin*.
- ***Underground storage and recovery (USR) —***similar to MUS; any type of project whose purpose is the artificial recharge, underground storage, and recovery of project water.
- ***Vadose zone well—***a well constructed in the interval between the land surface and the top of the static water level and designed to optimize infiltration of water.

Many additional technical terms and abbreviations may be found in the Glossary.

SOURCES: Bouwer (1996); State of New Mexico, 2001, Available online at *http://www.ose.state.nm.us/doing-business/ground-water-regs/ground-water-regs.html*; Well Abandonment Handbook; Dillon (2005); Municipal Water District of Orange County, available online at *http://www.mwdoc.com/glossary.htm*; Arizona Department of Water Resources: Underground Storage and Recovery Regulations, available online at *http://www.azwater.gov/dwr/Content/Find_by_Program/Wells/WellAbandonmentHandbook5.pdf*; WRIA Watershed Management Project, available online at *http://www.wria1project.wsu.edu/watershedplan/WMP_Master_Glossary.pdf*.

BOX 1-2
What's in a Name?

While the concepts and practices of recharge, storage, and recovery of water have existed for many years, the terms used to describe them are varied widely, and have changed over the years. In determining the terms to use as part of this study, the committee reviewed existing terms (see Box 1-1). Some of these terms, such as infiltration ponds, describe only the recharge method. Others, such as Aquifer storage transfer and recovery (ASTR), refer to single-purpose wells whereby recharge occurs in one well and recovery occurs in a downgradient well. Aquifer storage and recovery (ASR) generally refers to dual-purpose recharge and recovery wells. Other terms, such as Arizona's Underground Storage and Recharge (USR), were coined by legislatures or regulatory agencies in developing laws and rules to describe a range of activities. In Australia and other counties, management of aquifer recharge (MAR) describes intentional banking and treatment of water in aquifers (Dillon, 2005).

At the risk of adding another term to a crowded field, the concept of "Purposeful recharge of water into an aquifer system for intended recovery and use as an element of long-term water resource management" requires its own phrase. For this, the committee selected *managed underground storage of recoverable water* (MUS). This term is slightly different from the original term developed in the creation of the study, which was sustainable underground storage of recoverable water. The rationale for the selection of this term is as follows:

Managed captures the idea that these systems are deliberately and intentionally developed and operated to meet specific objectives while preventing or mitigating adverse impacts on human health and the environment. While committee members supported the concept of the development of these systems in an economically, physically, and environmentally sustainable manner, a consensus existed among the committee that the term "sustainable" could not be specifically defined within the broad context of this report. The term "managed," however, implies the existence of a manager, or project proponent, who is accountable for the development and operation of the system, with oversight by regulatory agencies.

Underground storage refers to the deliberate placement of water into an underground location through a recharge method, which could include surface infiltration and percolation through the vadose zone to a saturated aquifer or placement directly to an underground location in a saturated aquifer. The committee has described the operation of vadose zone wells in the report, but has found few successful systems to evaluate for physical, water quality, and institutional factors. The term "storage" also implies that the manager of the project intends to recover the water for a particular use—as opposed to systems where the intent of the recharge is primarily to prevent land subsidence, control saltwater intrusion or movement of contaminant plumes, or generally raise groundwater levels.

Recoverable water reinforces the concept that the water is being stored with the intent of recovery for a particular use. The ultimate use of the water to be stored impacts the ways in which the system is developed, operated, and regulated, particularly when reclaimed water is the source water.

The committee hopes that the acronym MUS will become a useful and well-understood addition to the water management lexicon.

The number of MUS projects is increasing rapidly. In 1983, there were three operating aquifer storage and recovery (ASR) systems in the United States. By 1994 there were 22 of these recharge well projects, and as of late 2005, there were about 72 systems in operation (Figure 1-2), with approximately 100 more in development (Pyne, 2005). These are located not only in the arid southwest-

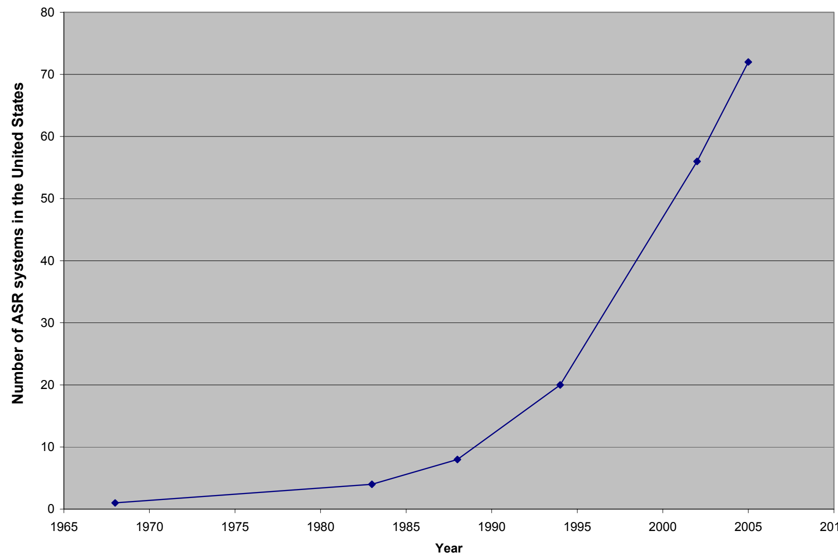

FIGURE 1-2 Growth of aquifer storage and recovery systems in the United States, 1968-2005. SOURCES: Pyne (1994, 2002); Pyne, ASR Systems, written communication, December 11, 2005. Reprinted with permission from Pyne (2005). Copyright 2005 by Pyne.

ern United States and in the Atlantic Coastal Plain areas, but also in the Pacific Northwest and even in the Midwest (Figure 1-3). The nation's oldest ASR system is employed by the seaside resort community of Wildwood, New Jersey (population 5,436) and the technology is being considered for use by New York City (population 8,000,000). In the Florida Everglades, more than 300 wells have been envisioned to recharge up to 3.8 million m^3 of water per day for ecological use, flood control, and water supply (USACE and SFWMD, 1999), while the Southern Nevada Water Authority currently has the largest ASR wellfield intended primarily for potable water supply, with more than 50 wells. At the other end of the spectrum, many small coastal towns along the Atlantic recharge water seasonally in small, one-well ASR systems to limit seawater intrusion and store water for the summer tourist season (AWWA, 2002). Suburban communities in Oregon, Washington, and Colorado are developing underground storage capacity, rather than relying on agreements with larger cities that possess surface storage facilities to meet their growing water demands.

Recharge through surface spreading has also grown increasingly common since early attempts in the late 1800s and is now employed in major metropolitan areas. For example, alluvial aquifers in Los Angeles County and the Santa Ana River watershed have been recharged through surface spreading of local river water, imported water from other watersheds, and recycled water. Today such managed recharge provides a majority of groundwater replenishment in

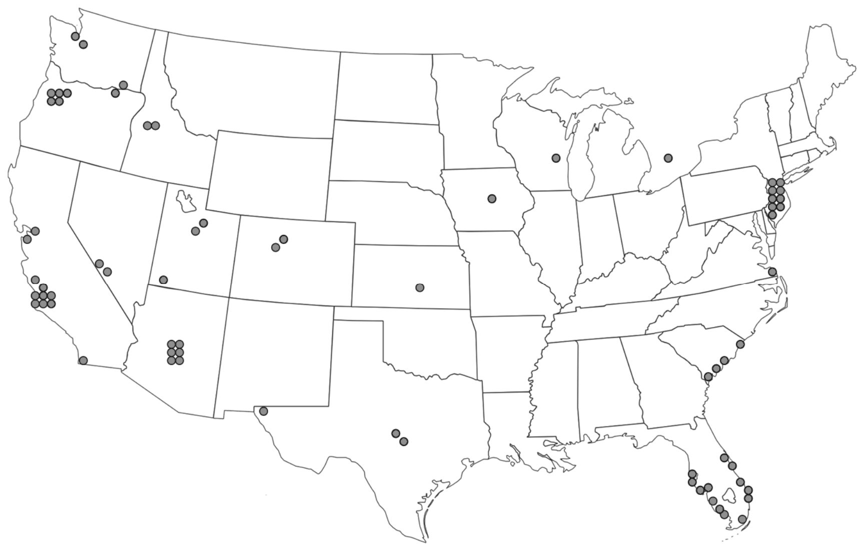

FIGURE 1-3 Distribution of aquifer storage and recovery systems in the United States, 2005. SOURCE: D. Pyne, ASR Systems, written communication, December 11, 2005. Reprinted with permission from Pyne (2005). Copyright 2005 by Pyne.

Southern California. Groundwater basins in this region support a population of more than 15 million people.

In Orange County alone, managed recharge of more than 300 million m^3 of water per year offsets the pumping demands on the Orange County groundwater basin, which provides well over half of the water needs for 2.3 million residents (*http://www.ocwd.com*). The principal wholesale water agency in the region, the Metropolitan Water District of Southern California, has developed storage agreements in several groundwater basins to provide additional supplies for drought years and emergencies. The Orange County Water District is currently constructing the largest indirect potable reuse facility in the world, which will provide 88 million m^3 of highly treated recycled water per year for recharge using both wells and surface spreading. Other projects to store water underground are in operation or in development for many areas of the Southwest, including the rapidly growing communities of Las Vegas and Phoenix. In short, MUS has become a widely accepted tool in water managers' portfolios—although, in areas of the country where this approach has not yet been applied extensively, it may still be perceived as experimental or impractical.

Despite the growing utilization of MUS and its many successes, there remain many questions about the conditions under which one's proposed goals can be achieved and the consequences of the use of MUS systems at large scales. Mineral transformations that occur during storage are poorly understood, as are the conditions under which inorganic or organic chemical contamination problems may be either improved or exacerbated. The long storage times associated with underground aquifers suggest that the consequences of these projects–either beneficial or detrimental—will also be long-lived.

In addition to questions about the physical, chemical, and biological aspects of MUS, the widespread interest in using MUS to address water supplies raises the question of whether existing water institutions are positioned to manage the long-term and widespread consequences of such systems or to facilitate the most effective strategies. A novel technology can be a challenge for water laws and institutions that have existed for decades. Some jurisdictions have responded with specific statutory schemes that facilitate the review and implementation of MUS projects. In other areas, regulatory hurdles still greet new MUS project proposals. Interjurisdictional issues are not uncommon, since aquifer boundaries are rarely aligned with institutional boundaries. Distinct laws govern the same water before, during, and after recharge, leading to uncertainties as to how current water rights laws might apply. Ownership and responsibility when recharged water moves in the ground, or causes perturbation of surrounding water supplies, may be unclear. Current regulation of aquifer storage systems is in the early stages of development in many parts of the country.

Interagency project regulation is also often an issue, since MUS systems represent uniquely interrelated concerns of groundwater protection, water supply and water resources management and (if the system is used to store water intended for potable use) drinking water. Where wells are used for recharge, the federal Underground Injection Control (UIC) program applies to MUS projects. The UIC program is implemented directly by the U.S. Environmental Protection Agency (EPA) in some states and by state agencies in others. States may have their own water quality standards, over and above federal requirements, that must be followed to protect groundwater and, in some instances, drinking water supplies. Some states have developed formal procedures for review of project permit applications to involve various water quantity and quality regulatory agencies, as both state and federal agencies to streamline the regulatory and permitting process and define agency roles. Still, ensuring that management of MUS systems is performed in a balanced approach that addresses water use, groundwater protection, and drinking water regulatory concerns can be a challenge.

The growing interest in underground storage of water raises the need for a better understanding of MUS. There are now enough operational systems that information on long-term performance in a range of geologic and hydrogeologic environments is available. These technologies will clearly be used even more widely in the future, and an ability to evaluate the likely success of a proposed system with some accuracy is critical.

Based on this, the Water Science and Technology Board organized a planning meeting in Washington, D.C. in April 2003, cosponsored by the AWWA Research Foundation (AwwaRF), to assess the degree of interest in the topic, followed in time by this consensus study. A large number of institutions contributed financially to this report (see preface ii). The statement of task (Box 1-3) was primarily derived from feedback received during this planning meeting.

The report is intended to (1) provide an integrated assessment of physical, chemical, operational, and institutional issues; (2) identify gaps in the science and practice that limit our understanding and provide a prospective examination of how these gaps might be closed; (3) provide guidance to prevent development of systems founded on unsubstantiated assumptions or poorly conceptualized models; (4) improve the accuracy of predictions of system performance over time, especially with respect to plugging or dissolution of the aquifer; and (5) provide a scientific basis for monitoring plans to track performance of operational systems and to gain knowledge for the design of future systems. The report also discusses financial and economic considerations within the context of

BOX 1-3
Statement of Task

Note: the original statement of task used the phrase "Sustainable Underground Storage" in lieu of "Managed Underground Storage."

The proposed study will provide an overview of some of the research and education needs and priorities concerning managed underground storage technology and implementation. It will also assess geological, geochemical, biological, engineering, and institutional factors that may affect the performance of such projects, based in part on a review and evaluation of existing projects.

Specifically, the study will assess and make recommendations with respect to research and education needs on the following questions:

- What research needs to be done to develop predictors of performance for underground storage projects based on the character of the recharge water in terms of contaminants, disinfectants, and microbes, the hydrogeology and major ion geochemistry of the source water and the aquifer, and the well or basin characteristics?
- What are the long-term impacts of underground storage on aquifer use—hydraulic, geotechnical, geochemical, adsorptive capacity of contaminants—at wellhead and regional scales, and can these impacts be ameliorated?
- What physical, chemical, and geological factors associated with underground storage of water may increase or decrease human and environmental health risks concerning microbes, inorganic contaminants such as nitrite, disinfectant by-products, endocrine disruptors, personal care products, pharmaceuticals, and other trace organic compounds?
- Are there any chemical markers or surrogates that can be used to help assure regulators and the public of the safety of water for groundwater recharge? What should we monitor and at what spatial and temporal scales?
- What are the challenges and potential for incorporating managed underground storage projects into current systems approaches to water management for solving public and environmental water needs?
- How do the institutional, regulatory and legal environments at federal, state, and local levels encourage or discourage managed underground storage?

challenges and opportunities. Although economic impacts are important considerations in MUS project planning and management, a comprehensive discussion of the topic is outside the scope of this study.

To address the issues associated with MUS and meet the objectives in its statement of task, the committee met five times over a period from February 2005 to June 2006 in Washington, D.C. (twice), Irvine, California, Phoenix, Arizona, and Woods Hole, Massachusetts. The first four meetings were partly open session for information gathering and discussion; the final meeting was closed in its entirety. The committee reviewed and evaluated existing information, including that published previously in journals, consultants' reports, or presented orally at the meetings.

Chapter 2 further defines the concept of MUS systems (summarized briefly above), provides further information on the development and history of MUS systems and how they function, and identifies the major issues associated with MUS systems to be addressed in the subsequent chapters of this report. Chapter 3 examines hydrogeological factors that determine the feasibility of aquifer recharge, identifies knowledge gaps and research barriers in understanding hydrogeology of MUS, and outlines recommendations for further research.

Chapter 4 focuses on water quality of the source, aquifer, and recovered water, particularly as related to human health and the environment. Chapter 5 addresses economic, legal, and jurisdictional considerations of MUS systems. Chapter 6 has been included to address the management aspects of MUS systems, providing a review of the stages of an MUS project and examining some key operational issues including clogging, monitoring and indicators, public perception, and financial considerations. Finally, Chapter 7 presents MUS in an overall water resource systems context for the nation.

Within this structure, there are numerous cross-cutting themes. For example, monitoring of MUS systems is addressed as a general issue in Chapter 2, with more specific monitoring issues explored from hydrogeological, water quality, regulatory, and management perspectives in Chapters 3, 4, 5, and 6, respectively.

CONCLUSION

The challenges to sustaining present and future water supplies are great and growing. The present overdrafting of aquifers and overallocation of rivers in many regions is a clear indication of these challenges, but the former also creates in many cases the underground storage potential needed to accommodate MUS systems. Thus, demand for water management tools such as MUS is likely to continue to grow.

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2
Overview of Managed Underground Storage Systems

This chapter provides the background information necessary to understand the detailed science, technology, and institutional issues related to managed underground storage (MUS) systems that are presented in subsequent chapters. The chapter begins with an overview of the components of MUS systems, as they are used throughout the report, and briefly explains some of the issues associated with each component. A condensed history of the evolution of MUS systems, focusing on the development and use of these systems within the United States, follows. Next, a review of the types of uses for which MUS systems have been or are being developed, and of other drivers behind the development of these systems, including agency-sponsored programs, is provided.

COMPONENTS OF MANAGED UNDERGROUND STORAGE SYSTEMS

Throughout this report, MUS systems are discussed in terms of five major components:

1. Source of water to be stored
2. Recharge method
3. Storage method and management approach
4. Recovery method
5. End use of recovered water

Opportunities and issues related to the selection, development, use, and regulation of MUS systems are typically tied to these components, and subsequent discussions regarding hydrogeology and hydraulics (Chapter 3), water quality (Chapter 4), legal, regulatory and economic issues (Chapter 5), and management of systems (Chapter 6) are usually tied to one or more of these components. While issues related to water sources and end uses may be common to both underground and surface storage of water, many of these issues are unique to underground storage systems, such as the potential interactions between the stored water and the native water in the surrounding aquifer.

Figure 2-1 illustrates some of the categories of issues encountered in MUS systems. The words in italics represent some of the criteria associated with each component that affect system selection and design. Note that many MUS systems contain some form of pretreatment before recharge and posttreatment during recovery. Monitoring of the stored water is often required. A source of

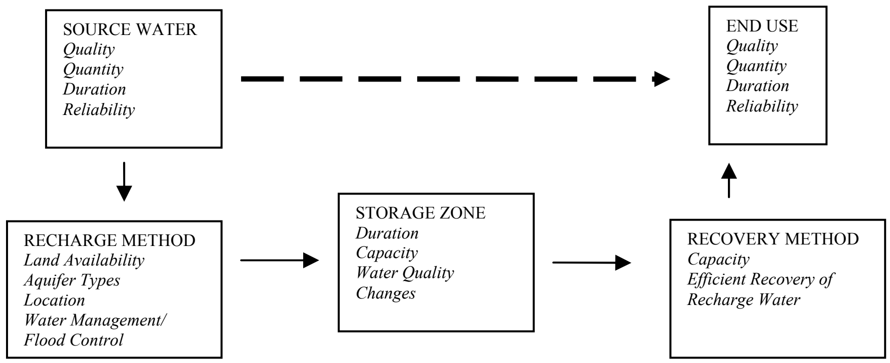

FIGURE 2-1 The five major technical components of MUS systems and some major design criteria.

water is required for all systems, but selection of the source is tied to the end use (particularly with respect to whether that end use is to be potable or not), as are the treatment and management during recharge, storage, and recovery. Major factors that impact the selection of recharge methods include aquifer type, land availability, and proximity to the water source.

These and other factors are described in the sections that follow.

Source Water

A variety of source waters may be used for underground storage, such as surface water, groundwater, stormwater, treated effluent, and (rarely) produced water. Waters from difference sources may have very different water quality characteristics. The water source used for recharge depends on availability, quality, duration, and reliability, as well as regulatory constraints.

When considering the end use of the water, a suitable water quality source must be selected. However, variations in source water quality and quantity may be mitigated during storage provided adequate storage time and capacity are available. Water quality improvements may occur during pretreatment prior to recharge, during storage, and during posttreatment prior to use. Ideally, the selection of source waters will minimize pre- and posttreatment requirements since these increase overall system cost. Pretreatment may be required to maintain infiltration rates, prevent negative interactions with aquifer materials, and prevent degradation of existing groundwater quality. Pretreatment requirements for recharge basins may be as simple as a stilling basin to remove heavy loads of solids prior to application. Stormwaters and surface waters are typically applied to recharge basins without pretreatment.

High quality source waters such as treated drinking water may suffer from water quality deterioration during storage, although subsurface storage may also provide protection for drinking water quality. Water quality sources such as reclaimed water typically tend to improve during subsurface storage since there is a large potential for improvement.

When reclaimed water is used and the final use is drinking water, the MUS system is referred to as indirect potable reuse (IPR; NRC, 1998) and special pretreatment or post-treatment requirements often apply to ensure that drinking water standards are not compromised and the receiving aquifer is not contaminated. When IPR systems use recharge basins, conventional water reclamation technologies are often sufficient to prevent significant deterioration of existing groundwater quality and water quality improvements are observed during subsurface transport. When IPR systems use recharge wells, advanced treatment technologies such as reverse osmosis are often used to prevent clogging and deterioration of groundwater quality.

Stormwater is often captured in retention basins that serve the dual purpose of capturing it and recharging it into the ground. Stormwater quality and quantity can be highly variable, and consistent with the National Research Council (NRC, 1994) report caution is needed in determining that the water is of acceptable quality for recharge. As noted in Chapter 4, limited data exist on the use of stormwater for MUS, and research is needed to determine the true potential of this little utilized but potentially important source water.

Produced water is a by-product of oil and gas production, and its disposal water is often a problem. Recharge of the produced water for future recovery is an option, provided the water quality of the receiving aquifer is not compromised. However, produced water is usually not suitable for placement in drinking water aquifers due to high salinity and the presence of organic contaminants, and it would generally require extensive treatment prior to recharge.

Recharge Method

The major methods that have been developed for accomplishing recharge are through recharge basins or through wells. With recharge wells, dual-purpose aquifer storage and recovery (ASR) wells may be used for both recharging and recovering stored water, or the water may be recovered through a separate well. Although not as often used for MUS systems, subsurface infiltration methods such as vadose zone wells can also be applied in unconfined aquifers, combining some of the advantages of both surface recharge and well recharge. Figure 2-2 illustrates the difference in location of recharge basins, vadose zone wells, and recharge wells with respect to the saturated zone of an aquifer.

The selection of recharge method will depend on aquifer type and depth and aquifer characteristics, which impact the ability to recharge water into the storage zone and recover that water later. The use of recharge basins and vadose zone wells is restricted to unconfined aquifers, while direct recharge and ASR wells may be used in both unconfined and deeper confined aquifer systems.

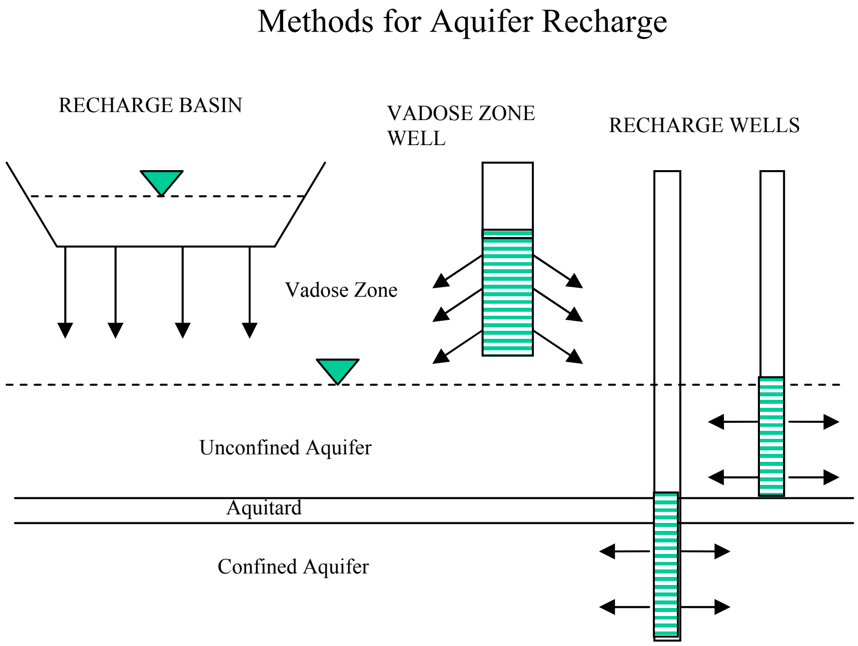

FIGURE 2-2 Major methods for aquifer recharge. Recharge basins are the most common type of surface spreading (see Box 1-1), which includes recharging water at the surface through recharge basins, ponds, pits, trenches, constructed wetlands, or other systems. Consistent with the figure, recharge wells can be used in either confined or thick, unconfined aquifers. Vadose zone wells are the least common of the methods shown.

Regulatory considerations may also come into play; for example, U.S. Environmental Protection Agency (EPA) Underground Injection Control (UIC) regulations regarding nonendangerment of groundwater are a particular concern for systems using well recharge (see Chapter 5).

Subsurface infiltration methods have been used to protect recharged water from evaporation losses or contamination, or where the land surface is not suitable for surface infiltration due to lack of land ownership and control, pavement of land surface, or other land uses that may cause surface infiltration methods to be infeasible. If there is a relatively thin layer of impermeable soil or man-made land cover (e.g., pavement) above the aquifer or vadose zone and the aquifer or vadose zone is relatively close to the land surface, an infiltration pit, trench, or shaft may be used. If the impermeable layer is thicker or farther below the land surface, recharge wells must be used.

Pretreatment requirements for recharge basins to maintain hydraulic capacity are not extensive since the accumulation of biological growth and suspended solids in the upper layer of soil is expected. Hydraulic capacity is normally maintained by drying and scraping to remove the clogging layer near the soil surface. Extensive pretreatment is generally required for vadose zone wells to prevent biofouling and clogging from solids.

Vadose zone wells are a relatively new technology and no effective techniques have yet been demonstrated for backwashing or cleaning them after they have clogged. Therefore, clogging must be prevented through the use of careful pretreatment.

Direct recharge wells must also be treated to prevent biofouling and clogging from solids. Introduction of a disinfectant (e.g., chlorine) may be effective at preventing biofouling. Reversal of flow in direct recharge wells may prevent the accumulation of solids and mitigate problems with biofouling; the ability to develop and minimize fouling of dual-purpose wells is one of the advantages that have been identified for ASR wells.

The hydraulic capacity of recharge basins depends on the local soil characteristics and the clogging potential of the water to be recharged. The capacity of vadose zone wells depends on the hydraulic conductivity of the vadose zone soils. Similarly, the hydraulic capacity of recharge wells depends on the characteristics of the receiving aquifer. Vadose zone wells have hydraulic capacities that are comparable to recharge wells. A hectare (2.5 acres) of recharge basins might be equivalent to a single recharge well. Therefore, the extensive land requirements for the use of recharge basins have made wells a more popular choice for groundwater recharge in urban areas. The cost of wells depends primarily on the depth of the well while land is the primary cost associated with recharge basins.

The selection of recharge method is a key consideration in determining the costs, issues, and operation and maintenance requirements for the MUS system. Some of the key characteristics associated with each recharge method are summarized in Table 2-1.

Storage Zone

The one component of MUS systems that may not be effectively engineered is the actual aquifer system used for storage. The capacity of the aquifer to store water is one of the most critical factors in selecting a site for underground storage systems. A second consideration is water quality improvement and/or deterioration that may occur during storage as a consequence of complex biogeochemical reactions. A third is impacts on the aquifer, such as clogging of aquifer pore spaces.

Recharge water may be stored in confined and unconfined aquifers. Other methods of subsurface storage, such as underground caves or abandoned mines, have also been used but are not considered in this study, which focuses on

TABLE 2-1 Major Characteristics of Aquifer Recharge Methodologies

	Recharge Basins	Vadose Zone Wells	Recharge Wells (including ASR)
Aquifer type	Unconfined	Unconfined	Unconfined or confined
Pretreatment requirements	Low/minimal technology	Prevention of clogging and biofouling	Prevention of clogging and biofouling
Estimated major capital costs US$	Land and distribution system	$100,000-250,000 per well	$100,000-1,000,000 + per well
Capacity	1000-20,000 M^3/ha-d	1,000-3,000 m^3 per well	2000-6000 m^3 per well
Maintenance requirements	Drying and Scraping	Drying and Dis-infection	Disinfection and flow reversal
Estimated life cycle	>100 years	5-20 years	25-50 years
Location of aquifer-water contact	Vadose zone and Saturated zone	Vadose zone and Saturated zone	Saturated zone

storage in an aquifer. Selection of an appropriate storage zone is an important consideration, impacting costs, the physical ability to get water into and out of the storage zone, and the potential for water quality impacts (negative and positive) on both the storage and the native water. Specific aquifers may also be protected by regulatory programs that must be considered in selecting and managing MUS systems.

During storage in unconfined aquifers, the groundwater table may rise and distinct mounds of water may develop below recharge basins or vadose zone wells. While increasing groundwater levels is often a goal of groundwater recharge, rising groundwater levels may have negative impacts if landfills or structures are located adjacent to groundwater recharge facilities. Storage in confined aquifers will increase the pressure in the aquifer, but actual groundwater levels may rise only if the aquifer is partially unconfined.

ASR wells provide the ability to store water in aquifers with poor-quality water such as brackish aquifers. The ASR wells produce a zone of stored water around the well that displaces poor-quality water and creates a storage zone of high-quality water. The stored water may be recovered from the storage zone, and the recovery efficiency (see "Recovery Efficiency and Target Storage Volume," Chapter 3) depends on the blending of injected water with the existing poor-quality water. The efficiency of recovery may be highly variable depending on the hydrogeology of the aquifer system, and the efficiency will determine the economics of such a system.

During aquifer storage, the time that water is stored in the subsurface is controlled by the design of the recharge method and recovery system. Systems that contain separate recharge locations and recovery locations often have defined flow paths and residence times between the point of recharge and the point of recovery. However, in the case of ASR systems, the last water to be recharged will likely be the first to be recovered and the residence time of stored water is highly variable. Consequently, if water quality changes occur during ASR, the water quality changes may also be variable. Recovery wells are located primarily for practical reasons such as proximity to point of use or conveyance system. When separate recharge systems and recovery wells are used, the recovery wells do not necessarily recover the same water that was recharged, however, the stored water remains available for future use. After the water is recovered, treatment prior to use may be required depending on the specific use requirements.

During storage, a variety of water quality transformations may occur depending on biogeochemical processes. Transformations because of changes in redox conditions and chemical interactions often occur rapidly and may impact the hydraulic capacity of recharge wells in addition to changing water quality. Transformations that depend primarily on biological reactions such as the biodegradation of organic compounds often occur slowly, and longer storage times are often necessary to achieve the full effects of water quality transformations during storage. Aquifers consisting of alluvial materials such as sand and gravel have a large amount of surface area that may contact the water traveling through the aquifer. This surface area mediates many biogeochemical reactions that may improve water quality during subsurface transport. Fractured and karst aquifers may have flow paths through fissures and conduits where surface area contact between the water and aquifer materials is limited. Since most biogeochemical reactions are surface mediated, water quality transformations that occur in alluvial materials may not be expected in nonalluvial aquifers where preferential flow paths exist.

During storage, both water quality improvements and deterioration may occur (Chapter 4). Improvements often come from the same natural processes that attenuate naturally occurring contaminants; many groundwaters do not require any treatment for potable purposes. Water quality improvements that are microbially mediated, such as the biodegradation of organic compounds, tend to correlate with longer storage times. When water is left in an aquifer for a long time it may approach native groundwater quality.

Water quality deterioration often occurs due to geochemical interactions resulting from redox changes as the injected water and the native aquifer water mix and continue to flow through the subsurface. The resulting dissolution of minerals may cause inorganic contamination of the recharged water. Water quality deterioration is most commonly associated with recharge wells for several reasons. Chlorination of the injected water to prevent biofouling is often necessary, and the resulting disinfection by-products may persist and/or actually increase in concentration during subsurface storage. Also, when water is injected, there are rapidly changing velocities as water moves away from the point of recharge, and if redox gradients occur, the potential for widespread geochemical interactions with negative consequences exists.

When recharge basins are used for groundwater recharge, chlorination is not necessary unless it is a regulatory requirement, and disinfection by-products have not been observed to be a problem with recharge basins. While redox changes may occur below recharge basins as a consequence of wetting and drying cycles, these changes occur slowly and rapidly changing velocities are not associated with most recharge basins. A plume of recharged water below recharge basins may become anoxic if sufficient oxygen demand was present in the recharged water. This commonly occurs with bank filtration systems in Europe, and the recovered water must be treated for dissolved iron and manganese. Since bank filtration systems do not have a vadose zone, there is no opportunity for aeration of the water during subsurface transport while most recharge basins have some opportunity for aeration during vadose zone transport.

Development of redox gradients is not the only potential cause of water quality changes during storage. Dissolution and precipitation reactions are caused by chemical differences, most commonly differences in the acidity or alkalinity of the waters. Salinity differences can also lead to interactions between the water and the aquifer materials. Chapter 4 discusses water quality issues in detail.

When water is stored in an aquifer, there is considerable uncertainty about the flow path of the stored water and the potential changes in water quality. These uncertainties may be reduced by monitoring (Chapter 6). Monitoring of flow paths may be accomplished by measuring water levels and/or pressures, and the measurements may be input into groundwater flow models to assess groundwater movement. Monitoring flow paths is relatively inexpensive, however, and capital costs will increase as the depth of required monitoring wells increases. Monitoring water quality transformations requires obtaining water samples and analyzing the samples either on-site or in an analytical laboratory. The level of difficulty and cost depends on the type of sampling equipment required and the analyses that must be performed. When emerging contaminants such as pharmaceuticals and personal care products are of concern, the costs for analysis of a single sample may exceed several thousand dollars. For recharge basins and vadose zone wells, suction lysimeters are necessary to obtain vadose zone samples. It may take more than 24 hours to obtain a vadose zone sample and sample volumes often limit the analyses that may be completed. Especially for projects using reclaimed water where monitoring requirements are stringent, monitoring may be the largest cost associated with the project.

Recovery Method

The location of recovery wells may affect several key factors associated with underground storage systems. Recovery wells may be located to direct the recovery of stored water toward proximity to the final use of the stored water. For systems that use recharge basins or vadose zone wells, the screened depth of the recovery well can have an important impact on storage time. As water percolates through the vadose zone, it accumulates in the uppermost portion of the aquifer and travels primarily in the horizontal direction under saturated conditions. By locating the screened interval below the top of the aquifer, the storage time before recovery can effectively be increased since vertical groundwater velocities are typically orders of magnitude lower than horizontal groundwater velocities. Of course, a well pumped in this zone would be continuously drawing antecedent groundwater, so such a strategy would not be appropriate for aquifers containing brackish or saline water.

For direct recharge wells, the screened interval does not tend to affect travel time since the primary component of flow is horizontal unless the recharge and recovery wells are very close or the recharge zone is very thick. For dual-purpose wells, the recharge and recovery well are the same, resulting in variable storage times. Land ownership and zoning are also considerations impacting the location of recovery wells.

Not all recovery occurs with wells. Along the Platte River in Colorado, water is taken from the river during high-flow, low-demand periods to offset impacts of well withdrawals from alluvial aquifers on more senior surface water rights. The water is placed in recharge ponds or ditches during nonirrigation seasons, where it seeps into the aquifer. It then flows in the subsurface back to the stream, which "recovers" it directly as seepage. Box 2-2 describes this creative system in more detail.

End Use

The final use of recovered water is the most important factor driving the economics of MUS systems, as well as many of the decisions regarding site selection; recharge and recovery methods; timing and duration of storage; available options for source waters; requirements for pre- and posttreatment; and permitting and regulatory constraints. This includes type of use (e.g., drinking water, irrigation water, industrial cooling water, environmental water) and timing of use (e.g., long-term storage for emergency use vs. operations to address seasonal variations in water availability and demand).

The final use of the stored water will dictate the desired final water quality. In many cases, treatment prior to the final use is primarily to prevent undesirable interaction with the distribution systems. For example, if potable reuse is desired, disinfection is often a standard practice to prevent biofilm development in the distribution system. When stored water is used for irrigation, posttreatment is typically unnecessary because the water quality is often comparable to or better than alternative surface water supplies with respect to pathogens and solids.

HISTORY OF MANAGED UNDERGROUND STORAGE SYSTEMS

Groundwater recharge for the purpose of storing water for future use has a very long history, with examples of surface recharge for water storage dating back over millennia. In the KaraKum Plain Desert of Turkmenistan, layers of clay with low hydraulic conductivity hold water at shallow depths underneath sand dunes. Consequently, nomadic tribes in Turkmenistan were known to dig trenches radially from sand dunes. The trenches were graded toward the dunes to collect rainwater that could be stored below the dunes. This simple form of groundwater recharge allowed the water to be stored for future use by simply excavating the sand dune (United Nations, 1975).

Bank filtration systems have been employed dating back to the nineteenth century. During bank filtration, river water is extracted indirectly by drawing it through the subsurface prior to use. While bank filtration systems do not provide storage of surface water underground, they do demonstrate the potential for water quality improvements during subsurface transport. Some bank filtration systems have been in operation for over 100 years (Grischek et al., 2002), and although they are not defined as MUS systems, many of the data on water quality transport during bank filtration are applicable to other underground storage systems.

During the twentieth century, advances in the science of groundwater hydrology led to the integration of deliberate and managed storage of water supplies underground into the development and integrated management of water supplies for various uses. While many of the issues associated with surface recharge and well recharge systems are similar, the technologies have evolved somewhat separately. The histories of surface recharge and of well recharge are presented separately below.

The history of groundwater recharge for the purposes of underground storage of water to be recovered for later use is closely tied to the history of other types of artificial recharge to conserve or enhance aquifers, prevent saltwater intrusion, induce bank filtration, prevent land subsidence, or other purposes.

History of Surface Recharge MUS Systems

Many underground storage systems have consisted of recharge basins where excess surface waters were retained and allowed to percolate to a receiving aquifer. The use of recharge basins was a logical extension of flood retention basins where excess drainage waters in urban areas were stored. Since there was no use for the water stored in the retention basins, subsurface storage would prevent water losses from evaporation and allow the water to be used in the future.

When floodplains were used for locating recharge basins, the underground storage systems also provided the benefit of diverting floodwaters and maintaining floodplains. The use of recharge basins is limited primarily to storage of water in unconfined aquifers, where no impermeable layer separates the recharge basin surface from the aquifer. Recharge rates can be enhanced by various means. Often times, the ground is excavated to increase percolation rates by removing less permeable surface soils. In addition to recharge basins, pits, trenches and shafts may be excavated for purpose of enhancing the recharge of unconfined aquifers.

In the United States, attempts at artificial recharge began in the late nineteenth and early twentieth centuries. Many of these projects were oriented less toward augmenting groundwater supplies than toward draining surface water for agriculture. However, there were exceptions. For example, water from Mill Creek and the Santa Ana River in Southern California was used to recharge the Bunker Hill Basin beginning in the 1890s and 1911, respectively (California Regional Water Quality Control Board, 1995). In nearly all cases, this recharge water was untreated. Therefore, success was most commonly achieved in highly porous and permeable aquifers such as limestones and fractured basalts where bacterial growth and suspended sediment deposition had relatively less impact (Weeks, 2002).

Long Island, New York, and Southern California were the foci of more scientific efforts to use artificial recharge to conserve or enhance groundwater storage beginning around the 1930s. For example, stormwater runoff collection basins were built on Long Island to collect water and permit it to infiltrate to the unconfined aquifer. Their number has increased from 14 basins in 1950 to more than 3,000 today (Ku and Simmons, 1986). The first large-scale planned operation of groundwater recharge using municipal wastewater in the United States was implemented by the Sanitation Districts of Los Angeles County in 1962, using secondary effluent as source water and recharging via recharge basins (NRC, 1994). Artificial recharge in the fast-growing State of Arizona did not begin at a large scale until the Granite Reef Underground Storage Project was permitted in 1994.

History of Recharge Wells

As groundwater withdrawal and water supply problems became more critical in the twentieth century, techniques to store water in confined aquifers using recharge wells were developed. Interest in using wells for groundwater recharge specifically to store water supplies increased after World War II, tied in part to concerns regarding potential attacks on water supply facilities. The U.S. Geological Survey (USGS) was involved with a number of early well recharge investigations with western cities, including Walla Walla, Washington (Price, 1960); Salem, Oregon (Foxworthy, 1969); Portland, Oregon (Brown, 1963); and Amarillo, Texas (Moulder and Frazor, 1957). Price (1960) noted that the use of untreated surface water with high suspended solids (2 mg/L total suspended solids) resulted in significantly degraded well efficiencies. Walla Walla, Salem, and Portland have all since developed operational aquifer storage and recovery facilities using basaltic aquifers in the same vicinity as these early well recharge experiments (Shrier, 2004).

The need to control seawater intrusion into aquifers motivated the development of recharge well systems to provide a hydraulic barrier between seawater and inland freshwater aquifers. In Orange County, California (see Box 2-1), Water Factory 21 began injecting water into the coastal barrier in 1976. Several alternative sources of water were evaluated for the recharge program including imported water, deep well water, reclaimed municipal wastewater, and desalted

BOX 2-1
CASE STUDY: Orange County Water District, Fountain Valley, California

The Orange County Water District (OCWD) began pilot studies in 1965 to determine the feasibility of injecting effluent from an advanced wastewater treatment facility into aquifers in the Talbert Gap at the mouth of the Santa Ana River to create a freshwater mound that prevents seawater intrusion. The 15 million gallons per day (Mgal/d; 57 x 10^3 m^3/d) facility, known as Water Factory 21 (WF-21), began recharge of treated wastewater in 1976 via 23 multiple-cased recharge wells. Additional wells have been constructed in recent years. WF-21 operated from 1976 until 2004, when it was decommissioned to begin construction of a new water purification system.

WF-21 received activated sludge secondary effluent from the adjacent Orange County Sanitation District (OCSD) Plant No. 1. The treatment processes at WF-21 changed through the years. In its final configuration, it consisted of microfiltration, reverse osmosis, and advanced oxidation using hydrogen peroxide and ultraviolet radiation. Although originally intended as a seawater intrusion barrier, the bulk of the injected water flows inland to augment groundwater used as a potable supply source. Extensive monitoring at WF-21 verified that the treatment provided is capable of producing water that meets all regulatory requirements for indirect potable reuse, including those related to xenobiotics and other trace organic contaminants.

In the 1990s, OCWD estimated that an additional 45 to 70 Mgal/d (170 x 10^3 to 265 x 10^3 m^3/d) could be recharged using existing recharge basins in the Orange County recharge area in Anaheim and Orange. A recharge project called the Groundwater Replenishment (GWR) System was conceived by OCWD and the Orange County Sanitation District to provide a new reliable drought-proof water supply, prevent seawater intrusion, improve groundwater quality, reduce ocean discharge, and defer the need for a new ocean outfall. In the first phase of the project, 70 Mgal/d (265 x 10^3 m^3/d) of purified water will be used for recharge. The GWR System is expected to become operational in November 2007. The source water and treatment processes for the GWR System will be the same as those used in WF-21's final configuration. The majority of the treated water will be pumped approximately 14 miles (23 km) through a 78-inch (198-cm) pipeline through the Santa Ana River corridor to Kraemer Basin in Anaheim, one of the deep recharge basins used in the Orange County inland recharge area. Some of the water, 15 to 40 Mgal/d (57 x 10^3 to 150 x 10^3 m^3/d) depending on time of year, will be diverted to an expanded Talbert Gap Seawater Intrusion Barrier previously served by WF-21.

The estimated capital cost of the GWR System is $480 million, and the estimated annual operating and maintenance cost is $22 million.

seawater. The water supply selected for recharge was a blend of deep well water and reclaimed water. The creation of hydraulic barriers with recharge wells could be done in both confined and unconfined aquifers. The development of hydraulic barriers protected an important source of groundwater from contamination while simultaneously replenishing the existing groundwater supply with a source that would have been discharged to the ocean. As discussed in this case study (Box 2-1), this system began primarily for seawater intrusion prevention, but evolved to include groundwater replenishment as well.

ASR systems, in which wells designed for the dual purpose of both recharge and recovery of water were integrated into a water supply system, were also being developed around this time. ASR systems were attractive to water supply agencies because an existing distribution system could be used for both water supply and storage. For example, a surface water supply could be treated to drinking water standards at a surface water treatment plant and distributed for both direct use and aquifer recharge. The first ASR system was implemented in Wildwood, New Jersey in 1968. In the late 1960s, California passed its State Water Plan with significant plans for underground storage of water, to be imported from Northern California to Southern California, through artificial recharge. Sites were subsequently developed in California in the 1970s. The earliest use of ASR in California was at the Goleta Water District, operational since 1978. Other early users of ASR in California include sites operated by the City of Oxnard and the City of Camarillo, both of which began operations in the late 1970s. New Jersey, California, and Florida (whose first ASR well was in Manatee County in 1978; Pyne, 2005) continued to be the only states with operational ASR facilities through the mid-1980s. In all three states, ASR was used as a form of water storage, but ASR facilities in these states were often also used as a tool for groundwater management in aquifers that were experiencing declining water levels and saltwater intrusion.

During the 1990s, the City of Scottsdale, Arizona, embraced the concept of using vadose zone wells for groundwater recharge. As real estate prices increased toward the end of the twentieth century and appropriate locations for surface recharge basins became scarcer, the need to develop a cost-effective method to recharge deep unconfined aquifers led to the development of vadose zone wells. Vadose zone wells are essentially shafts that are engineered to inject water efficiently into the ground. In Scottsdale, direct recharge wells would have to be 500 feet deep and vadose zone wells were determined to be economical even if the life cycle of the wells was only five years. The vadose zone wells were relatively inexpensive compared to direct recharge wells, did not require the extensive space of recharge basins, and could be placed in a variety of locations. The Scottsdale experience is also an example of where overdraft has created a tremendous storage zone.

REASONS FOR USING MANAGED UNDERGROUND STORAGE

The development of various MUS methods and the increasing prevalence of MUS systems are driven by increasing demands for water in general, as well as the advantages that can be gained through use of underground storage versus surface storage. As in any water planning exercise, the use of MUS systems for storage of water supplies is compared with other alternatives for storage (e.g., surface reservoirs or tanks) and reduction of demand (e.g., various conservation measures).

In all cases, economics is an important consideration for the selection of any project, although there may also be several other drivers causing project proponents to consider underground storage. For example, groundwater withdrawals may be restricted or prohibited unless a project proponent has first recharged that aquifer so that there is little or no net withdrawal (e.g., capacity use area laws in some eastern Coastal Plain states). There may also be environmental drivers or other public benefit objectives such as ecosystem restoration (Everglades, south Florida) or maintenance of minimum instream flows for salmon (Washington County, Oregon), brown trout (Squaw Valley, California), or other aquatic habitat. Underground storage may also provide secondary benefits related to maintenance of aquifer integrity and quality, such as helping to prevent aquifer dewatering or saltwater intrusion.

More detail on the types of uses that have developed for MUS systems, and the associated issues and constraints, are discussed later in this report. To illustrate the range of applications of MUS systems, however, a few examples of types of uses are provided below.

Seasonal Water Supply

Many MUS systems are developed to take advantage of seasonal availability of water supplies and seasonal demand, most often for municipal water supply, although seasonal irrigation demands are also often a consideration. In a recent survey of ASR facilities, more than half of the facilities surveyed operated their systems primarily for seasonal water use (AWWA, 2002).

Multiyear Water Storage or "Water Banking"

Several MUS facilities have been developed to provide multiyear storage in case of drought. Dependent upon site conditions and expected losses to the aquifer during storage, permitting requirements may reduce the amount of stored water that can be recovered if storage occurs for more than one year. As with seasonal storage of water supplies, MUS systems do not have the evaporation losses of surface water supplies and also require less use of land surface space for water storage that is needed only in drought situations.

Emergency Water Supplies

In addition to storing water for drought, MUS systems have been developed to provide water supplies when surface water storage facilities or treatment plants are impacted by more catastrophic events. For example, Walla Walla, Washington, uses MUS as a means of protection from forest fires. Recent catastrophic forest fires in the West during the current drought have caused water stored in surface reservoirs to become unusable due to increased sedimentation from post-fire erosion. MUS systems have been developed to ensure water supplies in case of hurricanes in South Carolina and floods in Iowa. MUS systems have also been cited as a means of backup storage of water supplies if there are impacts on treatment plants or surface water storage and distribution systems from earthquakes, brownouts, and terrorist attacks.

Availability of Water Rights

Both seasonal and long-term water availability may be tied to the *prior appropriation* legal system used throughout the western states. While the most senior water rights are typically for mining, followed by agriculture, and cities, the greatest population growth is occurring in the newest suburbs with the most junior water rights. Suburban municipalities in these and other growing western counties may have the financial resources to buy or lease agricultural water rights, but changing the use on these water rights (i.e., moving water from the farms to the cities) can be legally difficult and politically sensitive. The periods of peak demands for municipal uses, during the summer months, are also the periods when agricultural demands, with the more senior water rights, are greatest.

Older, more established large cities with existing storage and treatment facilities (e.g., Denver, Los Angeles) may be willing to develop agreements with newer suburban communities to provide water during wet periods, but they are likely to focus on their own service areas during periods of shortage. If adequate groundwater storage zones are available, MUS systems provide a means by which municipalities with junior water rights can develop storage facilities relatively quickly to capture water during seasons when those junior water rights are available, or when water rights transfers and exchanges can be arranged with more senior water rights holders, and to recover that water during periods when more junior water rights would be called out. This may be especially important in the future, because in some areas the allocation of water rights has been based on overoptimistic hydrological forecasts and junior water rights holders may be left without water for long periods of time (NRC, 2007).

Species Recovery and Habitat Protection Programs

Management of water resources to meet species recovery program requirements, particularly for endangered species, has been a major concern for water managers. In the Pacific Northwest, MUS systems have been developed to reduce stresses on surface water flows and stream habitat during low-flow periods by enabling water users to recover stored water in lieu of using surface water rights. One MUS system, in Walla Walla, Washington, has also developed a voluntary experimental project to take cooler water that had been stored in the aquifer and place it directly into Mill Creek during low-flow periods, when fish can be impacted by high stream temperatures, as part of a species recovery effort for the endangered steelhead salmon (Shrier, 2004). The largest ASR system currently under development is part of the Comprehensive Everglades Restoration Plan (CERP) in Florida. Box 2-2 provides a case study of the use of surface recharge systems in Colorado to provide both recharge and habitat benefits.

Groundwater Resources Management (Water Levels and Water Quality)

Several MUS systems have been integrated into regional efforts to manage groundwater levels and groundwater quality. By maintaining water levels by offsetting pumping with recharge, rather than mining nonrenewable groundwater resources, water users can reduce well interference and pumping costs, as well as prevent aquifer dewatering, land subsidence, and other impacts from stresses to groundwater resources by withdrawals. Arizona has an aggressive groundwater resources management program and uses ASR to recover groundwater levels in a stressed aquifer. As part of this program, ASR systems in Arizona are required to leave 5 percent of the recharged water in the aquifer. ASR has also been used to prevent potable groundwater from being impacted by saline water or contaminant plumes. The Equus Beds Aquifer Storage and Recovery project in Wichita, Kansas, is also being designed to control movement of a saline plume in addition to providing future water supply. The system will indirectly divert water from the Little Arkansas River through wells completed adjacent to the stream when flow in the river exceeds baseflow (*http://ks.water.usgs.gov/Kansas/studies/equus/*). As noted earlier (Box 2-1), some California ASR facilities have been located to help prevent seawater intrusion.

Industrial and Cooling Water Supply

Another use of MUS systems that has developed is for industrial applications. Micron Technology in Boise, Idaho, has been operational since 2001 and uses an MUS system to store surface water for use at a large semiconductor manufacturing operation. The facility owner-operator has cited the benefits of subsurface storage as a method for ensuring a more consistent water temperature

BOX 2-2
Case Study: Recharge Ponds for Streamflow Augmentation in Colorado

Recharge ponds for streamflow augmentation have been used in Colorado to control streamflows, offsetting the impacts of well withdrawals from alluvial aquifers on more senior surface water rights. Water is taken from the stream during high-flow, low-demand periods, when unappropriated water is available, and placed in a recharge pond or leaky ditch during nonirrigation seasons. The water seeps from the pond (or ditch) into the aquifer and flows back to the stream at a rate determined by the properties of the aquifer. There is a lag time in the impacts of both the recharge ponds and the well withdrawals. Typically, as allowed by the Colorado State Engineer's Office (SEO), the stream depletion caused by a well, or stream accretion created by a recharge pond, is calculated by the "Glover Method" (Glover, 1954), which is represented graphically in the Lower South Platte by the U.S. Geological Survey (Jenkins, 1968) and is referred to as the stream depletion factor (SDF).

Colorado water law prohibits the use of wells in alluvial aquifers unless there is a streamflow augmentation plan in place to offset the impacts of well withdrawals on surface water rights. The owner or operator of the pond (or ditch) receives augmentation credits that can be used against impacts to surface water flows from well withdrawals. These credits can also be leased to other water users whose groundwater use requires augmentation, if the pond is located and operated so that it can offset impacts of the well. The development of new recharge ponds accelerated rapidly during the 1980s and 1990s in response to emerging legal and administrative issues related to the development of permanent decrees and plans to augment streamflows to offset well depletions.

Habitat partnership programs have been involved with the development of habitat at managed groundwater recharge sites in Colorado since the mid-1990s to develop recharge ponds that also provide benefits for species habitat. The Tamarack Plan Recharge, Minnow Stream & Wetland Habitat Project, a demonstration project on the west side of the Tamarack Ranch State Wildlife Area, is one of the first sites at which a recharge facility has deliberately been designed and operated to maximize both the recharge credits produced for stream augmentation and the habitat benefits for wildlife. This project, developed as part of the Colorado Tamarack Plan, was designed and created cooperatively by South Platte Lower River Group water resources engineers, Colorado Division of Wildlife aquatic and habitat biologists and geomorphologists, and Ducks Unlimited ecologists.

There have been a number of recent projects developed in the Lower South Platte of Colorado in which habitat biologists and water resources engineers have worked together to design multipurpose facilities. Typically, habitat partnership programs develop agreements for conservation easements with the private landowners in this region. Landowners who are developing recharge ponds, and are interested in working with a habitat partnership program and designing the recharge ponds to provide habitat benefits, will typically contact the habitat partnership program to determine whether their sites would meet the eligibility requirements of that program.

and water quality than is typically found when using surface water supplies. MUS is also being explored as a means of storing water for industrial cooling purposes. In addition to the other benefits associated with MUS as a means of water storage for water users, industries that need cooling water can withdraw water from underground storage that is at a lower temperature than surface water and thereby use that water for cooling purposes at lower costs than would be incurred if warmer water were used. After that cooling water has been used, exchanges can then be developed with agricultural water users, who may prefer warmer water for use on some crops.

ROLE OF REGULATION AND FEDERAL AGENCY PROGRAMS IN MUS SYSTEM DEVELOPMENT

In some states, the lack of legal or regulatory mechanisms to address issues related to permitting of MUS systems was a hurdle that needed to be overcome before these projects could be completed. The development of regulatory programs for permitting and oversight of MUS systems facilitated MUS development in these cases. State programs designating certain aquifers as protected from new withdrawals, where MUS systems could be used to achieve a net zero impact or even a net increase in groundwater levels, also have led to increased use of MUS in some areas. In addition, there have been agency-sponsored programs, such as federal agency demonstration projects and research programs, have also played an important role in the development of MUS systems.

Since the 1980s, several states have developed laws or rules specifically addressing some aspect of MUS, particularly related to ASR systems. In states such as Oregon and Washington, development of some ASR facilities was delayed while new regulatory programs were being established, after which ASR development accelerated rapidly. Oregon and Washington each have more than a half-dozen ASR sites in operational or pilot stages. Arizona also has multiple ASR facilities that developed following the creation of its regulatory program.

Several MUS systems developed in eastern Coastal Plain states in response to the designation during the 1960s, 1970s, and 1980s of regions where net groundwater use was restricted to prevent saltwater intrusion, land subsidence, well interference, or other negative impacts. MUS systems could be used to enable well withdrawals during high-demand periods, such as for tourism in the New Jersey shore. Areas with restricted net groundwater withdrawals in New Jersey, Virginia, North Carolina, and South Carolina have all seen development of MUS projects.

Widespread use in the western United States of various forms of MUS (through both well recharge and surface recharge) was spurred by the U.S. Bureau of Reclamation's (USBR) High Plains States Groundwater Demonstration Project. This program was begun in response to concerns regarding falling groundwater levels in the High Plains (also known as the Ogallala) Aquifer and to calls for additional water supplies and water management following droughts in the late 1970s and early 1980s.

The original High Plains State Groundwater Demonstration Program Act was passed in 1983 and amended to include consideration of projects from all of the 17 western states in the contiguous United States that fall under the purview of USBR programs, rather than being limited to those states overlying the High Plains Aquifer. A total of 14 projects received federal funding under this partnership program, out of 42 originally proposed. In selecting the projects to be included in this program, USBR considered not only physical aspects of the sites, but also economic, institutional, and legal factors, to ensure that there was a sponsor that could meet cost-sharing requirements and that funding and project development would not be delayed by legal or regulatory impediments.

A wide range of aquifer recharge approaches were used at the different sites participating in the USBR program, including land use management, surface infiltration, recharge wells, and ASR. Some of the projects were intended for general groundwater replenishment, with no consideration of subsequent uses of the recharged water, while other projects recharged aquifers that were used primarily for municipal, industrial, or agricultural uses.

The largest federally sponsored ASR project–and by far the largest ASR project in the world–is that associated with the Comprehensive Everglades Restoration Program. The original restoration plan (USACE and SFWMD, 1999) proposed about 330 ASR wells, each with a capacity of about 5 Mgal/d (a total capacity of 1.65 billion gallons per day). They would have an average annual storage capacity of about 570,000 acre-feet and would represent about 26 percent of the new storage capacity for the restoration project (NRC, 2005). Five ASR pilot projects, located in different regions of South Florida, are planned or under way to test the viability of ASR as a large-scale water storage component of the restoration effort.

CONCLUSION

MUS systems all have five components: (1) source of water to be stored; (2) recharge method; (3) storage method and management approach; (4) recovery method; and (5) end use of recovered water. Issues associated with each component are discussed in subsequent chapters of this report. These systems use water from a variety of sources such as surface water, groundwater, treated effluent, and occasionally stormwater. They recharge groundwater through recharge basins, vadose zone wells, direct recharge wells, and ASR wells. The water is stored in a wide spectrum of confined and unconfined aquifer types, from unconsolidated alluvial deposits to limestones and fractured volcanic rocks. Recovery typically is achieved through either discharge wells or dual-purpose recharge and recovery wells, but occasionally is achieved via natural discharge of the water to surface water bodies. Finally, the recovered water is used for drinking water, irrigation, industrial cooling, and environmental purposes. Some simple forms of MUS using surface recharge have been applied for millennia. MUS systems using well recharge have a shorter history but have been in use for more than four decades. There is, therefore, adequate experience from which to draw some general conclusions about the degree to which MUS systems are successful in meeting their stated goals and the challenges and difficulties that some of them face.

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3
Hydrogeological Considerations

Development of an aquifer conceptual model through appropriate characterization of the physical underground storage system is a critical step in the development of a sustainable managed underground storage (MUS) system. In addition, analytical and/or numerical models can also be developed to evaluate water flow and solute transport in the aquifer and assess its potential as an MUS reservoir. To design a storage reservoir, engineers and hydrogeologists must have a good understanding of the hydrological properties of the aquifers to be used for storage and of the associated hydraulics. In particular, a successful MUS system design is predicated on answers to the following questions about the aquifer physical system and its hydraulics (including factors affecting success as listed by ASCE, 2001; Bouwer, 2002):

- What are the spatial constraints of the aquifer (basin extent, basin depth, aquifer thickness, interlenses, other boundary conditions)?
- What geological units are available for storage, and what are the hydraulic properties of these units (hydraulic conductivity, porosity, storage coefficient) (e.g., confined or unconfined aquifer, specific yield or storativity, hydraulic conductivities/transmissivities and hydraulic gradients, degree of homogeneity and isotropy, hydrocompaction, interaquifer hydraulic connection)?
- What temporal variations will affect the system (seasonal, climatic)?
- What are the short- and long-term impacts of the MUS system on the aquifer matrix, groundwater flow, or surface waters?

Additional decisions about the MUS system that significantly influence, or are influenced by, hydraulic characterization or aquifer attributes include the following:

- Will the water be recharged through spreading basins, wells, or other methods?
- Will the stored water be recovered by neighboring production wells (single function), recharge wells (i.e., aquifer storage and recovery [ASR] wells), or through gains in stream baseflow?
- How much of the stored water is intended to be recovered?

Successful design also requires identification of the source of water to be recharged and the anticipated uses of recovered water, which are discussed in other chapters. Hydrochemical and biological processes critical to MUS system success are described in Chapter 4.

Factors that can preclude MUS development include low available aquifer storage; low hydraulic conductivity; high probability of clogging during recharge; anticipated loss of recharge water; anticipated degradation of water quality due to physical, chemical, or biological processes, and anticipated changes in patterns of potentiometric gradients that would adversely affect existing water supplies.

The significance of these factors must be considered on a case-by-case basis. Depending on the operational goals of the MUS system, some of these negative factors may be acceptable provided regulatory requirements are met. Addressed briefly in Chapter 6 and not covered here are operational issues that affect MUS viability.

This chapter reviews the status of knowledge on the hydrogeology of recharge, storage and recovery processes as they relate to MUS. The chapter includes discussion of the hydrological properties of the geological formation to be used for storage, the aquifer boundary conditions, recharge and recovery methods to be used, and potential impacts of the MUS system on the groundwater flow and aquifer integrity. In addition, knowledge gaps and research needs related to the hydrogeology of MUS systems are identified.

AQUIFER TYPES AND CHARACTERISTICS IN THE CONTEXT OF MUS SYSTEMS

A requirement for the success of an MUS system is a comprehensive understanding of the hydrogeological properties of the aquifer to be used for storage. An *aquifer* is a layer, formation, or group of formations of permeable rock or sediment saturated with water and with a degree of permeability that allows water to be withdrawn or injected (Fetter, 2001; Marsily, 1986; Lohman et al., 1972). Sand and gravel layers, sandstone, and carbonate rocks usually form aquifers. This section describes hydraulic and hydrogeologic properties of aquifers, including flow and storage characteristics, and discusses aquifer classification with emphasis on considerations that are important to MUS.

Aquifer Classifications

Aquifer classification is generally based on composition, degree of confinement, and geometry at local and regional scales. Each of these is described below.

Lithology (Composition)

There are 66 principal aquifers—that is, regionally extensive aquifers or aquifer systems that have the potential to be used as a source of potable water—in the United States (Maupin and Barber, 2005). Each principal aquifer is classified into one of five lithologic types: unconsolidated and semiconsolidated sand and gravel aquifers; sandstone aquifers; interbedded sandstone and carbonate rock aquifers; carbonate rock aquifers; and igneous and metamorphic-rock aquifers. The total withdrawals of fresh water from these aquifers were estimated at 93.3 million acre-feet (83,300 million gallon per day [Mgal/d]) for the year 2000 (Maupin and Barber, 2005). About 92 percent of the total fresh groundwater withdrawals were used for irrigation, public supply, and self-supplied industrial applications. Withdrawals from the unconsolidated and semiconsolidated sand and gravel aquifers, including the High Plains aquifer, Central Valley aquifer system, Mississippi River Valley alluvial aquifer, and Basin and Range basin-fill aquifers, accounted for 80 percent (or 62,400 Mgal/d) of total fresh groundwater withdrawal for the above listed uses. In 2000, carbonate rock aquifers, primarily from the Floridian aquifer system, igneous and metamorphic rock aquifers (primarily the Snake-River Plain aquifer), and sandstone aquifers (primarily from the Cambrian-Ordovician aquifer system) provided 8 percent, 6 percent, and 2percent of total fresh groundwater withdrawal, respectively, from all aquifers in 2000.

In the western United States, MUS activities have been conducted primarily within unconsolidated alluvial fan, floodplain, coastal plain, and inland valley deposits. However, in other regions, consolidated aquifers are also used for MUS, such as carbonate aquifers in Florida and fractured igneous-metamorphic rocks in the northwestern United States.

All types of aquifers have been used for ASR, but in general ASR is easier to manage in consolidated aquifers where the formation provides a competent well without the requirement for screen and gravel pack (Dillon and Molloy, 2006). Carbonate aquifers show offsetting effects of carbonate dissolution on well clogging (Herczeg et al., 2004), but as discussed later in the chapter may have problems with mixing of injected and native waters. Fractured rock aquifers, even low-yielding ones, have been used successfully for ASR (Murray and Tredoux, 2002) with injection rates in some wells exceeding airlift yields. Coarse-grained sand and gravel are also very suitable for ASR storage targets, but care needs to be taken with well construction and completion, to reduce as much as possible the concentrations of organic and colloidal material introduced into the well. Storage in fine-grained unconsolidated media is more problematic and requires water with very low nutrient and colloidal concentrations in order to avoid chronic and irrecoverable depletion of the specific capacity of the ASR well.

Table 3-1 summarizes properties of major types of aquifers. The shape and extent of these aquifer types is governed by the geological history of the region, including the depositional environment and subsequent deformation (if any).

TABLE 3-1 Properties of Major Types of Aquifers

Matrix Composition	**Confinement**	**Porosity Type**
Carbonate	C, S, U	Dual porosity—intergranular & joints, fractures, solution conduits
Unconsolidated and consolidated siliciclastic sediments	C, S, U	Intergranular
Fractured or jointed igneous, metamorphic	C, S, U	Joints, fractures
Fractured sedimentary rocks	C, S, U	Dual porosity—intergranular and fracture

NOTES: Confined (C), semiconfined (S), and unconfined (U) including water table and may or may not be perched.

Degree of Confinement

There are three aquifer conditions with respect to confinement: unconfined, semiconfined, and confined. Aquifer confinement affects or limits methods of recharge, storage, and recovery. Therefore, MUS system performance varies for these different aquifer conditions. Importantly, confined and semiconfined aquifers can be recharged only by wells. Unconfined aquifers can generally be recharged by either wells or by surface spreading methods.

.*Unconfined aquifers* allow flow of water from the land surface into the aquifer (i.e., recharge). Therefore, unconfined aquifers are naturally unprotected from contamination due to a lack of intervening low-hydraulic-conductivity units, known as confining layers between the land surface and the aquifer. Unconfined aquifers are also referred to as *water table aquifers* because the upper surface of the saturated zone is at equilibrium with the atmospheric pressure. This surface is called the *water table*, which often follows the land surface topography with variations due to recharge and boundary conditions. As a result, the water table may reflect hills, valleys, and plains. Localized recharge may also cause mounding. In very highly permeable aquifers the water table is more controlled by the presence of boundary conditions, such as lakes and rivers.

In general, unconfined aquifers receive more recharge in upland areas where precipitation infiltrates into the ground, as well as near water bodies where seepage occurs. Discharge from an unconfined aquifer to the ground surface in low-lying areas usually occurs at springs or the bottom of surface waters (Fitts, 2002). Therefore, groundwater in unconfined aquifers interacts with surface water via several points or areas of connection, (e.g. rivers, lakes, wetlands, springs, and along coastal zones). By observing the hydraulic gradient, one can determine if a water body is "gaining" or "losing." For example, a *gaining* *stream* is recharged by the aquifer, whereas a *losing stream* discharges to the aquifer.

Unlike unconfined aquifers, *confined aquifers* are recognized by being isolated by a saturated or partially saturated low-hydraulic-conductivity, or "confining," layer on top of the aquifer. Rock or clay can form low-permeability barriers that impede or constrain the flow of water into and out of the aquifer. These confining layers allow pressure to build up in the aquifer system. An artesian well results when the pressure in a confined aquifer is sufficiently high that the groundwater in a well rises above the land surface. The water elevation in a well open to a particular point in a confined aquifer is known as the *piezometric head* at that point, which is the sum of the *pressure head* and the *elevation head* (Bear, 1988). The two-dimensional surface that is defined by mapping the head across the extent of a confined aquifer is the *potentiometric surface* or pressure surface.

Natural recharge zones where a confined aquifer becomes unconfined are important aquifer characteristics. In confined aquifers, these areas are created when the geological confining layers are absent, exposing the aquifer to infiltration. If a well is drilled in a confined aquifer, the water in this well will rise to the elevation of the recharge area.

Last, *semiconfined* or *leaky aquifers* are saturated aquifers underlying a low-permeability layer, or aquitard. The low permeability of the confining unit allows for limited recharge into and discharge out of this aquifer. The degree of confinement can vary with natural variability of the confining unit: composition (i.e., clay content), pinchouts, or localized discontinuities (i.e., breaches due to sinkholes or fractures).

Geometry and Scale

Conceptual knowledge of aquifer geometry at both regional and local scales is required in order to identify boundary conditions, which are important constraints on an MUS application. Aquifers within the hydrogeologic framework of a given region occur either closed or open basins.

An aquifer at the margin between the land and the ocean exemplifies an *open basin* condition. Open basins that reflect a broad shallow paleocoastal margin depositional environment for sediment deposition may contain sheet-like strata comprising the storage zones; hence, the lateral boundary conditions can often be considered infinite. On the other hand, vertical boundary conditions exert an important control on the behavior of the system in this hydrogeological setting, especially with regard to ASR.

If the anticipated storage formation is located in a closed basin, almost all of the recharged water can be retained within the basin except water lost through evapotranspiration in discharge areas. Most alluvial aquifers in the southwest United States, for example, are located in closed basins. These aquifers are surrounded by bedrocks and receive limited recharge from the mountain fronts or captured flow from the surface water system. Under natural conditions, water table slopes and groundwater movement will tend to conform to the surface topography. In many inland basins, this results in drainage from the basin at its lower end. Under such conditions, depths to groundwater will tend to decrease toward the downstream portions of these basins, particularly if there are geologic constrictions to reduce the rate of movement. If the water table intercepts the surface, discharge will occur either directly to surface water or as evapotranspiration via phreatophytes. This results in a loss of water from the basin. Should groundwater levels in these areas be drawn down as a result of artificial extraction, there will be a saving in the water that would otherwise be consumptively used by the phreatophytes. The value of water supply gained will need to be compared to the environmental values of the phreatophytes lost. With artificial recharge, water levels will typically rise, which can lead to increased discharge. As a result, the recoverable water may diminish as the length of storage time increases.

The storage zone geometry is also affected by local scale features and local variability (heterogeneity) in the hydrophysical properties of the aquifer. In sedimentary aquifers, the paleoenvironment in which the sediments were deposited affects the geometry of the storage zone. For example, if the storage zone is located with a paleofluvial (riverine) system, the geometry of the more permeable zones may be ribbon-like (Prothero and Schwab, 2004). In a mixed clastic-carbonate aquifer, storage zones may be more isolated both vertically and laterally than they are in a more homogeneous sandy alluvial aquifer.

Hydrogeological Properties

The hydrogeological aquifer properties that are most significant with respect to underground storage are the *hydraulic conductivity (or transmissivity for a confined aquifer)* and *storage coefficient (either specific yield or storativity)* (see text below and Glossary for definitions). Leakage from adjacent water-bearing zones (quantified through the *leakance*) also affects an underground storage reservoir. The geological processes that create the aquifer control the hydrogeologic properties that the aquifer possesses. For example, in aquifers comprising sedimentary rocks, the environment of deposition, depositional processes, and lithology (types of grains) affect hydraulic conductivity and storage properties through the spatial arrangements of and variations in the grain size and sorting, packing, roundness, and so on. Postdepositional processes such as compaction and cementation can reduce hydraulic conductivity while dissolution and fracturing tend to increase hydraulic conductivity.

Storage

The capacity of an aquifer to store water is described or quantified by the storage coefficient; *specific storage* and *specific yield* are the terms used for confined and unconfined aquifers, respectively. The aquifer properties that affect the specific storage are the total porosity and compressibility of the aquifer matrix. Specific storage ranges from less than 3×10^{-6} m^{-1} in rocks to 2×10^{-2} m^{-1} in plastic clays (Anderson and Woessner, 1992). *Storativity*, which is equal to the product of specific storage and aquifer thickness, defines the volume of water released from storage per unit decline in hydraulic head in the aquifer per unit surface area of the aquifer (Table 3-2).

The relationship between fluid pressure, effective stress, and flow is essential to understanding the mechanism of aquifer storage (Charbonneau, 2000; Fitts, 2002). Storage capacity is modified by compression or expansion in the soil or rock matrix as a response to effective stress. Effective stress is defined as the difference between the total stress and the stress supported by the fluid. The total stress is the weight supported by the surface divided by the surface area (Charbonneau, 2000). In other words, when pressures are lowered by removal of water during pumping, stress is transferred to the solid matrix and the solid matrix compacts as a result of the increased effective stress. When pumping ceases, water flows toward the area of reduced head, causing an increase in fluid pressure and a transfer of stress to the fluid phase. The reduced effective stress on the solid matrix causes an expansion of the matrix.

The specific yield quantifies the pore space that is drainable by gravity. In other words, it expresses the difference between the total water filled porosity and the water held by surface tension (i.e., undrainable water). Values of specific yield range from close to 0 for clays to more than 0.25 for coarse gravel (see Table 3-2).

There are two types of storage space used most commonly for MUS. One is the drained pore space within a geological unit; this space may have been created by historical groundwater withdrawal (i.e., groundwater overdraft or mining). In general, the available storage spaces in such depleted aquifers are laterally extensive and may have experienced a reduction in storage capacity as a consequence of consolidation or compaction of the aquifer matrix during historic pumping.

The second type of storage space is created by displacement of native water with recharge water creating a zone of freshwater around the recharge well (Figure 3-1). In other words, injecting freshwater into a confined aquifer will create an increase in the piezometric head commonly known as the “mounding effect” (e.g., Bouwer, 2002). An example of this type of storage would be an ASR well in a saline or brine aquifer. This type of storage space may be limited by available recharge area and/or by allowable pressures in the aquifer.

Porosity in an aquifer system changes throughout the geologic history of the media. The primary porosity, comprising, primarily intergranular space, is created during deposition in sedimentary rocks. It can be reduced by subsequent compaction and lithification. Secondary porosity is created through marked alteration of the original aquifer media. Examples include conduits formed by carbonate dissolution, partings along bedding planes, or fractures. The term "dual porosity" characterizes an aquifer that contains both primary and secondary porosity.

Hydraulic Conductivity and Transmissivity

Hydraulic conductivity describes the ability of the aquifer or any unit or volume within it to allow water flow. Hydraulic conductivity is dependent on the fluid (viscosity and density) and the geological medium (Viessman & Lewis, 2003). The dimensions of the connected water- filled pore spaces are the physical attributes of the medium that control the hydraulic conductivity. Hydraulic conductivity values can range over 12 orders of magnitude (Domenico & Schwartz, 1990). Low–hydraulic-conductivity values are indicative of a less permeable matrix such as clay or shale (confining units), while high values are indicative of a highly permeable matrix such as sand and gravel (Schwartz & Zhang, 2003). *Transmissivity* is equal to the product of the hydraulic conductivity and the aquifer thickness and is most often used in the context of confined aquifers. It thus quantifies the capability of the entire thickness of the aquifer to conduct water flow. Water also moves from one aquifer to another through a semiconfined or confined layer. *Leakance*, which is defined as the ratio of vertical hydraulic conductivity to the thickness of the confining unit or aquitard, was generally used to denote how fast or slow the confining unit may allow water pass through it. Table 3-2 summarizes ranges of these hydrogeological parameters, as well as storage parameters, from known MUS projects within common aquifer storage media.

The hydraulic conductivity of an aquifer can vary with location in the aquifer—termed *heterogeneity*—and/or with the direction of groundwater flow—termed *anisotropy*. The *Heterogeneity* and *anisotropy* of aquifer hydraulic properties must be known in order to plan an MUS system and develop accurate groundwater flow or solute transport models for such a system. The aquifer created in a fluvial sedimentary deposit provides an example of one that has heterogeneous hydraulic conductivity with lower conductivity in the finer-grained overbank or floodplain-generated units and higher values in the channel features. Heterogeneity in the hydraulic conductivity of aquifer storage units is the norm, rather than the exception. As discussed later in the chapter, heterogeneity often leads to a highly nonuniform distribution of water recharged by wells (Vacher et al., 2006)—not the subsurface "bubble" of stored water employed in simpler conceptual models.

Whereas heterogeneity indicates that hydraulic conductivity differs between points in an aquifer, anisotropy is the term that characterizes differences in hydraulic conductivity with direction of flow. Anisotropy can result in observations

TABLE 3-2 Approximate Hydrogeological Parameters in Aquifers Used for Underground Storage

Matrix Composition	Hydraulic conductivity (ft/day)	Transmissivity (ft^2/day)	Specific Yield	Storativity	Specific Capacity[1] (ft^3/day/ft)	Leakance (per day)
Carbonate	10^{-1} to 10^{3}	10^{2} to 10^{5}	0.01 to 0.1	10^{-3} to 10^{-5}	10^{3} to 10^{5}	10^{-2} to 10^{-5}
Unconsolidated and consolidated siliciclastic sediments	10^{-1} to 10^{2}	10^{2} to 10^{4}	0.1 to 0.3	10^{-3} to 10^{-6}	10^{3}	10^{-3} to 10^{-5}
Fractured igneous, metamorphic, and sedimentary rocks	10^{0} to 10^{-4}	10^{2}	0.05 to 0.1	10^{-2} to 10^{-5}	10^{3} to 10^{5}	-

SOURCES: Brown et al. (2005); Driscoll (1995); Leonard (1992); Pyne (2005); Reese (2003); Reese and Alvarez-Zarikian (2007); and Ward et al. (2003).
[1]An expression of the productivity of a well. It is defined as the ratio of discharge of water from the well to the drawdown of the water level in the well. It should be described on the basis of the number of hours of pumping prior to the time the drawdown measurement is made.

of order-of-magnitude differences in the vertical and horizontal hydraulic conductivities in a single core sample of aquifer material. Anisotropy contrasts are generally greater when vertical and horizontal flow directions are compared. Within a layered sedimentary system, for example, flow in the vertical direction is impeded by the presence of any low-hydraulic-conductivity layers, whereas flow in the horizontal direction may travel in laterally continuous, more permeable zones unimpeded by the low-hydraulic-conductivity layers. A massive (i.e., unbedded), very well sorted quartz sand or carbonate grainstone aquifer (i.e., nearly free of a clay-sized fraction) would be characterized as homogeneous and isotropic. On the other hand, a mixed siliciclastic-carbonate aquifer typical of the southeastern U.S. Coastal Plain would be considered heterogeneous and anisotropic.

In the context of aquifer storage, a dual porosity aquifer system can be considered a dual reservoir. While most of the water may exist within connected primary pore spaces through which water moves relatively slowly, water residing in the secondary porosity may travel at greater velocities (e.g., conduit flow in a carbonate aquifer). A prominent example of a dual- porosity unit that is frequently considered for MUS systems is the “Chalk” of England, which has up to 40 percent primary porosity, yet most of the flow is through fractures (Gale et al., 2002).

The scale of measurement strongly influences the resulting observations in dual-porosity aquifers. Because only the permeability of the matrix or primary porosity is captured in laboratory sample-sized measurements, much greater hydraulic conductivities are observed at the well-field scale where the volume of aquifer measured includes flow through the more permeable secondary porosity features. Fluid flow within secondary porosity can be non-Darcian including turbulent flow (high Reynolds number), and velocities may range from 10^2 to 10^3 feet per day, where these gravel seams or fractures are not continuous over large distances. Hydraulic conductivity values generally range from 10^{-3} to 10^1 feet per day in the less permeable (primary) counterpart of the dual-porosity system (Brown et al., 2005; Driscoll, 1995). Open basins and coastal plain aquifers that are comprised dominantly of dual-porosity carbonates are especially susceptible to issues of scale with regard to hydrogeological parameters.

Igneous and metamorphic rocks are generally not considered to have dual porosity because fracture porosity comprises nearly all of the open volume in which water can flow or be stored. Primary porosity in these comparatively brittle rocks is extremely low and rarely interconnected, unless the rocks have been significantly weathered. In a basaltic aquifer, zones of greatest hydraulic conductivity occur along lava flow boundaries; lava tubes comprise a unique type of secondary porosity.

Both groundwater modeling and effective monitoring design are facilitated by understanding the physical characteristics of the secondary porosity such as the size, orientation, and distribution of fractures or partings. The orientation of fractures and joints is generally related to present or paleo-stress fields; widening of these features may occur due to rock dissolution and mechanical breakdown. Conduit size is more dependent on the aquifer lithology (e.g., carbonate rocks dissolve more readily than silicic rocks) and history of exposure to chemically aggressive water.

Additional influences on the distribution of secondary porosity in carbonate rocks include changes in the position of the freshwater-seawater interface, sea-level fluctuations, climate change, and extensive pumping. Variations in lithology, depositional environment, and position of bedding planes also contribute to evolution of conduits that may yield complex flow systems.

Water Movement Between Aquifers or Between Aquifers and Surface Water

Aquifer Interaction

In an aquifer system, it is possible for water to move from a semiconfined aquifer of higher hydraulic pressure into an unconfined one or vice versa when the semiconfined aquifer hydraulic head is reduced by pumping. Water movement may also occur through windows or lenses between confined aquifers due to potentiometric head differences. Adding water to a confined aquifer can be accomplished only by increasing the pressure of water in already saturated pores (contrasted with the ability to add water to partially saturated pores above the water table in an unconfined aquifer) Interaction among aquifers at different physical elevations depends on the piezometric head between them and on the thickness, hydraulic conductivity, and integrity of the confining unit. Water from different aquifers may also be transferred through uncased wells or abandoned wells. Leakage between unconfined aquifers and semiconfined aquifers can be enhanced by increased head difference or reduced by decreased head difference as a result of recharge of one aquifer.

Surface Water and Groundwater Interaction

Groundwater commonly is connected hydraulically to surface water (Alley et al., 1999). In the natural system, the interaction takes place in three basic ways: a water body gains water from inflow of groundwater through its bed, through its margins, or via a spring or seep; loses water to groundwater by outflow in the same manner (seepage or sinkholes); or does both, gaining in some places and losing in others depending on local and temporal changes in hydraulics (seasonal or climatic changes affecting relative pressures). Groundwater-surface water interactions occur between aquifers and rivers, lakes, wetlands, retention ponds, infiltration trenches, and spreader canals. If the vertical gradient or the hydraulic conductivity is low, the flow rate between the water body and the aquifer is lower. Wells located closer to water bodies may have strong impacts on surface water flow, whereas distant wells tend to have lesser impacts. Pumpage of wells in close proximity to water bodies may greatly increase seepage, especially from coarse-grained stream channels or unlined canals and laterals.

These types of interactions are relevant to MUS projects because surface water bodies serve as boundaries that recharge or drain the aquifer. For example, water reuse projects could be implemented in coastal aquifers, where water delivered to canal systems that recharge the aquifer prevents saltwater intrusion from wellfield drawdown.

HYDRAULICS OF RECHARGE

As noted in the previous chapter, managed underground storage of recoverable water can be achieved using three different methods, namely surface spreading (e.g., recharge basins, modified stream beds, pits and shafts), vadose zone wells, and recharge or ASR wells, plus others including watershed management (water harvesting or enhancement of natural recharge). Each method is governed by its own hydraulics (ASCE, 2001; Bouwer, 2002; Pyne, 2005).

Surface Spreading

This method consists of releasing water from the source to a recharge basin, pond, pit, or channel for infiltration. This method of aquifer recharge can be used only for unconfined aquifers (see Box 3-1). It also requires large (land) surface areas to accommodate the recharge scheme that can also allow significant evaporation if infiltration is slow. Surface spreading usually requires both a diversion structure and an infiltration structure (ASCE, 2001).

Evaluation of infiltration capacity is critical for MUS because it dictates the method and size of the recharge site. The factors that affect infiltration capacity of artificial recharge projects include the composition of surface soils, the geology, subsurface hydrologic conditions, source water quality, and procedures used in the construction, operation, and maintenance of the recharge structure. The operational factors can ordinarily be managed to maintain favorable infiltration capacity. Therefore, the most important attributes to characterize the suitability of a recharge location for MUS are the soils, geology, and hydrogeology of the recharge location. Of particular importance are geologic structures or low-hydraulic-conductivity units that might form a barrier to groundwater movement and the position and hydraulic gradient of the existing water table or potentiometric surface.

Under certain geologic and hydrologic conditions, the groundwater mound developed as a result of spreading intersects the land surface. This can occur (1) when subsurface lenses with sufficiently low permeability exist that restrict the downward movement of the recharged water, creating localized mounds, and (2) when the water table is sufficiently close to the surface to cause a similar effect. In both cases, the infiltration capacity is essentially limited to the quantity of lateral flow from the mound, although under the first of these conditions there is probably a small amount of movement through the less permeable lens. The lateral movement of water away from the mound is generally found to be in substantial conformity to Darcy's law. Considerable literature exists presenting methods to estimate the shape and the rates of buildup and recession of groundwater mounds beneath recharge areas (Bouwer et al., 1999).

Artificial recharge by injection consists of using a conduit access, such as a tube well, shaft, or connector well, to convey the water to the aquifer (Figures 2-2 and 3-1). It is the only method to artificially recharge confined aquifers or aquifers with low-hydraulic-conductivity overburden. The water is recharged directly into the storage zone, and there are no transit or evaporation losses. This method can be particularly effective in highly fractured hard rocks and karstic limestone, but it is also used in unconsolidated or alluvial sediments.

BOX 3-1
Case Study: Capturing Water from the Santa Ana (California) River

The Orange County Water District (OCWD) in coastal Southern California is responsible for managing the underground water reserves that supply approximately 270,000 acre-feet per year from about 500 wells within OCWD's boundary. That quantity grows steadily, and projections indicate the demand may reach 450,000 acre-feet a year in the next quarter century (OCWD, 2006; *http://www.ocwd.com/_html/recharge.htm*). Groundwater reserves are maintained by a recharge system, which replaces water that is pumped from wells. OCWD facilities have a recharge capacity of approximately 300,000 acre-feet per year. About 2 million people depend on this source for about three-quarters of their water. Groundwater producers pump water from the groundwater basin and deliver it by pipeline to consumers.

Along a 6-mile long section of the Santa Ana River that belongs to OCWD, a system of diversion structures and recharge basins captures most of the river water that would otherwise flow into the Pacific Ocean. The district has 1,500 acres of land for use in its recharge program. The current average annual baseflow of the Santa Ana River is approximately 140,000 acre-feet. Storm flows add an average of 60,000 acre-feet per year, ranging from 10,000 to 500,000 acre-feet. The baseflow may increase by 100,000 acre feet over the next 20 years due to urban development in upstream areas. Increased urbanization creates more buildings and paved areas, which results in greater quantities of storm runoff. Population growth also causes a proportional increase in wastewater discharges to the river channel.

Anaheim Lake, one of OCWD's recharge basins. Photo courtesy of Orange County Water District.

Water flows down the Santa Ana River from Riverside and San Bernardino Counties, together with supplies imported from the Colorado River and from the California State Water Project. In the Cities of Anaheim and Orange in Orange County, a pattern of interlaced levees built of sand helps to slow the river's flow to maximize the amount of water that can percolate through the bottom of the river channel. Water is also diverted from the river into a series of recharge basins. These basins, with depths ranging from 50 to 150 feet, were formed in years past by sand and gravel mining operations. The soil along this stretch of the Santa Ana River is coarse-grained and sandy. Therefore, water readily seeps into sand and gravel layers below the ground surface. Groundwater is stored in the underground sand and gravel aquifers. Certain aquifers reach the surface in this recharge area of Orange County and can easily be recharged, while in other areas closer to the coast, a layer of dense clay overlies the aquifer and prevents efficient percolation of significant quantities of surface water.

continued next page

BOX 3-1 Continued

The district's deep recharge basins, such as Anaheim Lake, Warner Basin, and Kraemer Basin, gradually accumulate a thin layer of fine sediments and biological material that slows and can even stop percolation. Although the percolation rate in a newly cleaned deep basin can reach 10 feet per day, the rate can drop to nearly zero after six to eight months. Each of these deep basins is periodically emptied by means of submersible pumps, and the clogging layer is removed by scrapers or by a sand-washing device. Clogging affects only the upper 2 to 3 inches of soil. A twice-yearly cleaning cycle, which has replaced a single annual cleaning, increases percolation by as much as 40 percent.

Prevention plays an important role in solving the problem of clogging. A flocculation system at the Imperial headgates is being considered that could coagulate suspended solid particles so that they will settle out of the water as it passes through a series of desilting ponds. Three of these ponds help reduce the sediment load in water that is diverted to the recharge basins. This slows the formation of a clogging layer and thereby helps to maintain efficient percolation.

More than 100 species of wildlife are found on district lands, and OCWD cooperates with environmental organizations to preserve the natural habitat of these animals. Recreational opportunities include river trails for horseback riding, bicycling, and jogging; two recharge basins are also stocked for sport fishing.

Injection

Vadose Zone Wells

Vadose zone wells are boreholes (usually 10 to 50 m deep and about 1 to 1.5 m in diameter) in the unsaturated zone completed with a center pipe and the annual space between the pipe and the wall of the borehole filled with sand (ASCE, 2001). They are often used to dispose of storm runoff and to reduce flooding (also called drainage or recharge wells) most commonly in areas of relatively low rainfall.. A negative aspect of vadose zone wells is the introduction of contaminants that comes from recharge of untreated urban runoff (petroleum byproducts, metals, nutrients, pesticides, surface microbes). Pretreatment strategies are needed, including first-flush bypass, screens, filters, and disinfection systems. Such systems require assessment and monitoring of contaminant fate and transport during wet and dry periods. An important limitation of vadose wells is that there are no effective and reliable methods to reverse clogging.

Recharge Wells

Recharge of water into abandoned wells and wells specifically designed for artificial recharge has been practiced for many years with varying degrees of success. The use of recharge wells is confined largely to those areas where surface spreading is not feasible owing to the presence of low-permeability layers overlying the principal water-bearing deposits. They may also be more economical in metropolitan areas where land values are too high to utilize the more common basin, flooding, and ditch and furrow methods (ASCE, 2001).

Many attempts to recharge groundwater through wells in alluvial and sedimentary aquifers have yielded disappointing results. Difficulties encountered in maintaining adequate recharge rates have been attributed to silting, bacterial and algae growths, air entrainment, release of dissolved gases, rearrangement of soil particles, deflocculation caused by reaction of high-sodium water with soil particles, and chemical reactions between recharged waters and native groundwaters resulting in precipitates in the aquifer or well-casing perforations (Bouwer, 2002). However, the Los Angeles County Flood Control District, in California, has successfully operated recharge wells for many years, creating and maintaining a freshwater ridge to halt seawater intrusion in the Manhattan-Redondo Beach area in Los Angeles County. Favorable recharge rates have been maintained by chlorination and deaeration of the water supply and by conducting a comprehensive maintenance program on the wells.

The spacing of the recharge wells depends on the range of influence of a well, which in turn depends on the rate of water recharge, and on aquifer and well hydraulic properties, including the aquifer hydraulic conductivity, hydraulic gradient, length and diameter of perforated casing or screen penetrating the aquifer, and the open area of casing perforations or screen. In general, it has been found that gravel-packed wells operate more efficiently and require less maintenance than non-gravel-packed wells in alluvial aquifers. In addition, where water is being injected under pressure, it has been found that a concrete seal should be provided on the outside of the casing at a point where it passes through the relatively impermeable confining bed, to prevent the upward movement of water outside the casing.

For reasons covered in Chapters 4 and 6, long-term use of recharge wells in alluvial aquifers requires treatment of the injected water. Sediments must be removed completely. The clear water should be treated with chlorine, calcium hypochlorite, or copper sulfate to prevent the growth of bacterial slime and algae. The water must also be free of dissolved gases that may be released into the formation and cause air binding or air entrainment, which reduces permeability. In addition, care should be taken to ensure that in formations containing appreciable proportions of base-exchangeable clay, water containing a high percentage of sodium will not be used, since this will cause deflocculation of the aquifer sediments and rapid decrease in transmissivity. These treatment requirements may not be necessary in highly permeable limestone and volcanic aquifers. Gravity feed of stormwaters and treated wastewaters into the carbonates of the Floridan Aquifer System in Florida has been effective for many years with no evidence of significant plugging.

ASR Wells

An aquifer storage recovery well (Pyne, 2005) is designed in such a way that water can be injected through a single well and recovered through the same well at a later time. Recharge for ASR is similar to that for other recharge well systems except for the cycling of injection and pumping. With an ASR well or system, it is assumed that most of the recharged water will be pumped back because a relatively constrained zone of recharge water (i.e., a so-called bubble) will be retained until the stored water is recovered. However, aquifer heterogeneity and anisotropy, as well as density differences (if any) between the source water and native groundwater, tend to produce relatively amorphous shapes that describe the three-dimensional limits of the recharge water (e.g., bottle brush [Vacher et al., 2006], upside-down Christmas tree [Missimer et al., 2002]), or an "octopus" shape) that reflect preferential flowpaths in dual-porosity settings. In some of these cases, especially within a dual-porosity setting, injected water often cannot be fully recovered by the ASR well. Therefore, the term "bubble" in reference to the shape of the recharge water body could be misleading.

Casing, screen design, and storage interval should be determined by hydrological properties of the aquifer. Hydrogeologic constraints that are not assessed at the time of design and/or that change over time, such as the collapse of unstable geologic layers into the well borehole, may cause plugging and fouling. Naturally occurring and/or artificial fracturing, sinkholes, and karst terrain features may dictate where recharge water can flow and how much recharged water can be recovered. Poor well design and/or construction practices, including insufficient placement of grout; improper design of pumps, valves, and fittings; and excessive drawdown allowance can lead to low recharge rate, low storage capacity, and low recovery efficiency (see "Recovery Efficiency and Target Storage Volume" later in this chapter) (Bloetcher et al., 2005; Pyne, 2005). High recharge rates in wells can result in turbulent (also termed non-Darcian) flow around the well casing, which will impact well performance and water movement in the recharge zone. Incorrect injection pressures can also alter fracture networks by reopening existing fractures or generating new ones.

In most cases, it is easier to get a steady pumping rate through a production well than a steady recharge rate. Therefore, well designers tend to use pumping rates instead of injection rates to design recharge wells. Such assumption tends to overestimate the capacity of recharge wells, resulting in an underestimate of overall costs of the system. It was found, for example, that the recharge rate of an ASR well was approximately 50 percent to 70 percent of the pumping rate in El Paso, Texas (Boyle Engineering, 1999). The ratio of recharge to recovery specific capacity for comparable flows and durations typically ranges from 25 to 100 percent, with 50 to 80 percent being a reasonable range for unconsolidated aquifers (Pyne, 2005).

Despite the many possible complications, ASR has worked successfully in many locations. An example is given in Box 3-2. There are also cases, albeit many fewer, in which ASR has been attempted but abandoned (Box 3-3).

BOX 3-2
Case Study: ASR System at Boynton Beach, Southeast Florida

Among the 21 ASR systems in southern Florida (Brown, 2005), one of the most successful is the Boynton Beach East Water Treatment Plant located on the east coast in Palm Beach County. Compared to other ASR systems, this system has the highest recovery efficiency achieved per cycle (Reese, 2002) during its operation since 1992. Treated drinking water from the Wastewater Treatment Plant (WTP) is the source of recharged water. The ASR was constructed to recharge into the Hawthorn formation (i.e., sandy phosphate limestone), which is located within the upper Floridan Aquifer at a depth of 804 to 1,200 feet below land surface (Reese, 2002). The thickness of the storage zone's open interval at the Boynton Beach site is 105 feet, and transmissivity is reported to be about 9,400 square feet per day (CH_2M Hill, 1993). The zone of influence is at least 800 feet as reflected by observation from the monitoring well. The equilibrium pressure of the system is 10 pounds per square inch (psi) and there are no upward leaks. The maximum pressure range for recharge is 55-60 psi.

Recharge occurs during the wet season: June through December. After recharge has been completed, the pressure of the aquifer system drops from 60 psi to the natural value of 10 psi. One full cycle is defined as one wet and one dry season. Approximately 100 Mgal are recovered per cycle, while the pump itself moves 2.5 Mgal/d. A total of 24 separate recharge and recovery cycles have been completed with recovery efficiencies (i.e., the percentage of the total amount of potable water recharged for each cycle that is recovered) varying from 40 percent to 100 percent (Reese, 2002). Recharge-recovery cycles had been conducted for an average of about two cycles per year. Recovery efficiency seems to be linked to the length of the storage periods (Reese, 2002).

Water was recovered until the chloride concentration in the recovered water slightly exceeded 300 mg/L during the dry season. This is related to the fact that native groundwater had a chloride concentration of 1,900 mg/L. The goal of the water treatment plant is to have a chloride range around 70-80 mg/L. Recovered water has 250 mg/L of $CaCO_3$ due to the geology. Injected water has 40-50 µg/L of trihalomethanes, but when the water is extracted, these levels fall 3-5 µg/L, which is below the safe drinking water standard of 80 µg/L. Economic evaluation shows that the ASR alternative is considerably less expensive than other seasonal water supply options (Brown, 2005; CH2M Hill, 1993; Muniz and Ziegler, 1994). The main reason that the quality and amount of storage water are well maintained is the fact that the aquifer has a relatively low permeability zone located just underneath the confining unit.

Aquifer Storage, Transfer, and Recovery Wells

In aquifer storage, transfer, and recovery (ASTR), water is pumped from a different well than the alternate from the recharge well. This has been tested in Australia (Pavelic et al. 2004). In fact, it is very similar to the reclaimed water recharge systems in Orange County, California (Dillon et al., 2004) and El Paso, Texas (Sheng, 2005). In large projects, such as the Everglades Restoration, hundreds of what are nominally called ASR wells are planned. In fact, it seems unlikely that each well will only recover the water that it recharged. In such a case, ASTR would probably occur—whether planned or unplanned—and combined with ASR this might add to the overall efficiency of the system. In some

BOX 3-3
Case Study: ASR System at Taylor Creek (Lake Okeechobee), South Florida

The Taylor Creek ASR project site is located along the northeastern portion of Lake Okeechobee, in south Florida. It has generally been considered an unsuccessful ASR project due to its recovery efficiency (see "Recovery Efficiency and Target Storage Volume" later in this chapter) of only 15 to 36 percent on four tests in 1989 and 1991 (Reese and Alvarez-Zarikian, 2006).

The site was a South Florida Water Management District demonstration project to test the feasibility of storing large volumes of phosphorus-rich stormwater from Taylor Creek underground to prevent it from reaching nutrient-enriched Lake Okeechobee (CH2M Hill, 1989). The project was conceived as a single test well with an on-site groundwater monitoring well. The proposed storage zone in the upper Floridan Aquifer System is highly transmissive and porous fossiliferous limestone, and based upon test data and water quality sampling data, the ASR well was completed with an open-hole interval from 1,268 to 1,700 feet below land surface. The storage zone represented a confined leaky aquifer with a transmissivity of about 570,000 feet per day (CH2M Hill, 1989).

The source water from Taylor Creek was highly variable in composition, with total dissolved solids (TDS) ranging from 268 to 996 mg/L and total coliforms ranging from nondetectable to 7,500 per 100 mL. The ASR storage zone also had variable groundwater quality (TDS from 4,000 to 6,900 mg/L; CH2M Hill, 1989). The source water was treated prior to recharge.

Brown (2005) did an extensive analysis of the site's potential based on application of a newly proposed ASR planning decision framework. He concluded that the project seemed to be infeasible both technically and economically for each of three alternatives evaluated and recommended that the Everglades restoration program consider a more suitable project location if it is to do future testing of ASR in the area.

cases, recharge water that moves beyond the capture zone of an ASR well could be captured by a single-purpose recovery well.

Other Recharge Methods

Enhancement of natural recharge or watershed management offers an effective method to intercept dispersed runoff. Many water conservation techniques have been developed for hillslopes with the intention of preventing soil erosion and reducing surface runoff. These increase the infiltration and aquifer recharge. Traditional terraced agriculture is certainly one of the most common water harvesting methods in arid areas, particularly in the Near East such as Jordan (*http://www.ruralpoverty portal.org/english/learn/water/harvesting.htm*). Where the terraces are well maintained, they effectively control runoff and improve aquifer recharge, but once allowed to fall into disuse, they progressively lead to gully erosion, collapse of the retaining walls, destruction of the whole system, and severe modification of the hydrological regime. Therefore, whatever the economic virtues of such terraces, it should be recognized that their abandonment on a large scale can upset the hydrological conditions within a basin for a considerable period of time.

RECOVERY OF STORED WATER

Fate of Recharged Water

Will the recharged water remain in the aquifer until recovery occurs? In many cases, recharged water will migrate or mix with native groundwater due to hydrological and boundary conditions of the aquifer. If water is stored in shallow aquifers, stored water may flow back into a neighboring stream as baseflow or flow downward to recharge deep aquifers as the hydraulic head increases. As water levels increase in the shallow aquifer, evapotranspiration from the shallow aquifer may also increase. This section describes how processes in the aquifer affect recovery.

Differences exist between recharge basin systems and recharge wells with regard to these processes. In recharge basins (Figure 3-1(a)), the source water infiltrates into an unconfined storage zone through a recharge zone. Recharged water is depicted as a "mound" on native groundwater. The base of the lens is a mixture of recharge water and native groundwater, termed transitional water. The entire lens is the storage zone, whereas the vadose zone located above the water table is the recharge zone. The groundwater level will still fluctuate in response to changes in natural recharge, which is affected by changes in land use, seasons, and climates. Other factors include pumping of water wells and the influence of nearby recharge projects.

In vadose zone wells (Figure 3-1(b)), the source water is injected into an unconfined storage zone through a recharge zone. As in Figure 3-1(a), recharged water is shown as a mound on native groundwater with transitional water between the two.

In ASR wells (Figures 3-1(c) and (d)), the storage and recharge zones overlap. Recharge water has an irregular shape to reflect aquifer heterogeneity. Only in nonbuoyant, isotropic, homogeneous aquifer conditions would the term bubble appropriately describe the shape taken on by the recharge water. Proximal to the recharge water is the transitional water, in what is referred to as a buffer or mixing zone.

In any of the scenarios shown in Figure 3-1, the zones vary in size and geometry as a function of hydraulic gradient, dispersivity, presence of dual porosity, and relative density of the native and recharge water. If significant chemical differences exist between recharge water and native groundwater, the degree to which this mixing occurs depends on the *dispersivity of the aquifer*. *Mechanical dispersion* is a scale-dependent process that pertains to fluid mixing due to flow through heterogeneous media. *Diffusion* reflects the movement of dissolved species from higher to lower areas of concentration and does not require flow. Dispersive mixing will tend to increase with time and distance from the recharge well due to both molecular diffusion and "mechanical dispersion," which results from the heterogeneous nature of aquifers. In a dual-porosity storage zone, the role of diffusion is significant, whereas in a more homogeneous aquifer, the effects of diffusion are masked by dispersion.

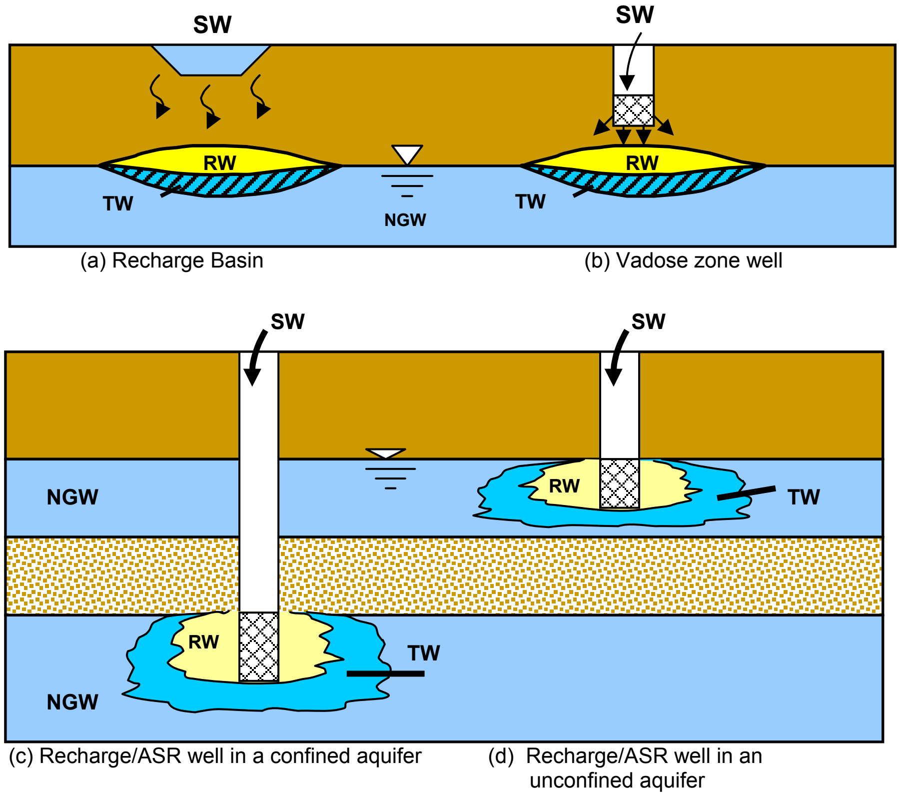

FIGURE 3-1 Mixing of waters with different recharge methods: source water (SW), either surface water, groundwater, or reclaimed wastewater; recharge water (RW); transitional water (TW, mixed recharge water and native groundwater); native groundwater (NGW).

Recovery Efficiency and Target Storage Volume

Recovery efficiency (RE) broadly reflects the proportion of recovered water in an ASR system and is an important concept in terms of MUS system performance. Variable applications of the term underscore the need to clarify its meaning. For example, RE has been defined as a fraction or a percentage and has been applied to describe usable water from individual cycles as well as the total or cumulative performance of a system. By some definitions, recovery efficiency may exceed 100 percent; however, many prefer to calculate RE in a manner that reflects recovery of the actual water injected into the aquifer through an ASR well. In this context, RE would not exceed 100 percent. From an MUS system management perspective, RE needs to be defined in terms of individual cycle tests and overall system performance. Therefore, the terms "cumulative recovery efficiency" (CRE) and "operational recovery efficiency" (ORE) are adopted herein.

Kimbler et al. (1975) define CRE as the ratio of the cumulative volume of fresh water injected minus the volume of unrecovered fresh water divided by the cumulative volume of fresh water injected. This definition yields a fraction and requires use of threshold value of a water quality parameter to determine the volume of recovered water that originated as artificial recharge. This limiting parameter should allow clear distinction between recharge and native water; examples include salinity, total dissolved solids, electrical conductivity, or a nonreactive tracer. The asterisk in Equation (3-1) reflects this limiting parameter. As an example, if salinity is used, the U.S. Environmental Protection Agency (EPA) drinking water standard for chloride (250 mg/L) is often the constraint (Reese and Alvarez-Zarikian, 2007). Pavelic et al. (2002) quantify this concept in terms of a "recovered mass" fraction. Modifying the Kimbler et al. (1975) definition to a percentage is more widely used in the industry:

$$CRE = 100 \times \frac{\text{cumulative volume of recharge water - cumulative volume of unrecovered recharge water}}{\text{cumulative volume of recharge water}} \tag{3-1}$$

As defined, the CRE reflects the overall recovery efficiency of the MUS system. For a given recharge and recovery cycle, ORE is applied in a similar form:

$$ORE = 100 \times \frac{\text{volume of recharge water during cycle - cumulative volume of recharge water not recovered}}{\text{volume of recharge water}} \tag{3-2}$$

Recharge and recovery of fresh water into a freshwater aquifer requires careful selection of the water quality parameter used to distinguish between the recharge water and the native groundwater, given that CRE and ORE should not exceed 100 percent. Operational recovery efficiency (Equation 3-2) is consistent with Pyne's (2005) definition of RE: "the percentage of the water volume stored in an operating cycle that is subsequently recovered in the same cycle while meeting a target water-quality criterion in recovered water."

Bear (1979) noted that a certain volume of recharge water—namely, that portion of the injected water body extending beyond the water divide for pumping—can never be recovered by the recharge well itself. However, this portion of recharge water can be partially recovered by pumping from downgradient neighboring wells (e.g., ASTR, Pavelic et al. 2004) or by simply discharging back into the surface water system. In addition, native brackish water, which otherwise may not be usable, can be recovered for beneficial uses after treatment or mixing with recharged water.

In an ASR feasibility study, Merritt (1985) investigated the effect of hydrogeological parameters on RE. He noted that "recovery efficiency [ORE] improves considerably with successive cycles providing that each recovery phase ends when the chloride concentration of withdrawn water exceeds established criteria for potability (usually 250 milligrams per liter), and that freshwater injected into highly permeable or highly saline aquifers (such as the boulder zone) would buoy rapidly." In this manner, recharge water mixed with native groundwater is left in the transition zone with successive cycles.

Recovery efficiency varies with recharge method, hydrological and hydrochemical properties of the aquifer, and recovery methods. Brown (2005) has done an extensive analysis of these factors. The dispersivity, thickness of the storage zone, preexisting groundwater gradient, recharge volume, rock type, presence of high-permeability zones, length of storage time, density of ambient groundwater relative to recharge water, ambient groundwater quality, and number of recharge and recovery cycles can all be important. The roles of transmissivity and anisotropy are still unclear; porosity does not appear to be a major factor (Brown, 2005). In general, surface recharge compared to deep injection in the same aquifer may have a lower recovery efficiency due to evapotranspiration and other losses related to surface spreading.

Maximized ORE is a long-term operational objective for any ASR system. To this end, Pyne (2005) recommends a "target storage volume" (TSV) approach to meet a predetermined recovery volume goal. The TSV is defined as "the sum of the stored water volume and the buffer zone volume in an ASR well" (Pyne, 2005). In the context of the physical hydrogeologic setting (Figure 3-1), the TSV is the sum of recharge and transitional water volumes. Implementation of the TSV concept involves one or more high-volume recharge phases intended to displace native groundwater early in the development of the system. This initial large-volume buildup is designed to develop a transition zone sufficiently far from the ASR well such that subsequent smaller cycle test volumes would minimally recover transition water. Ideally, successive cycles would yield increasing OREs approaching 100 percent to meet the targeted operational objective.

The rate at which the TSV is developed depends on the water availability, cost of water, aquifer hydraulic parameters, and regulatory issues (Pyne, 2005). For example, the rapid development of the TSV will likely maximize operational recovery efficiency and can be timed to minimize the cost of water or maximize water availability (off-peak demand periods).

Characterization of the hydrogeochemical system (Chapter 4) at a given ASR site prior to TSV development may facilitate progress along the path of regulatory authorization. For example, if water-rock-microbial interactions that may affect water quality are anticipated during recharge or storage based on experience from similar hydrogeochemical settings, the rapid development of a large storage and transition zone may inhibit the ability of the regulatory community to assess water quality changes at the ASR monitor well(s). Bench-scale studies, geochemical modeling, or initial smaller-volume cycle tests may be warranted. Sufficient time between each cycle test would allow completion of water quality analyses and data interpretation. Results of such an assessment could then be used to modify the next cycle test design (duration, pumping rate, volume, etc.) to optimize data collection and improve characterization of the hydrogeochemical process with the goal of mitigating its effects. Ideally an adaptive characterization method can move forward in parallel with operational strategies that improve ORE, such as the TSV concept.

A high hydraulic conductivity not only increases the potential for water to travel beyond the zone of recovery, but also promotes mixing with native groundwater. Conversely, low transmissivity may require extreme wellhead injection pressures during recharge and may cause excessive drawdown during recovery.

Different aquifer types result in marked differences in recovered water quality with time, as demonstrated by preliminary field data for ASR systems in Figure 3-2. Even at the earliest observation, the pumped water in the dual-porosity aquifers (fractured chalk and fractured sandstone) and, to a lesser extent, basalt and heterogeneous sandstone has the chemical signature of a mixture of the injected and native waters. When water is injected into a dual-porosity system, it flows through the secondary porosity much more rapidly than through the primary porosity. Water in some fraction of the primary pores is not replaced by recharged water. During storage, solutes in the primary and secondary pore spaces will move toward equilibrium through diffusion. The figure shows that when native groundwater quality is poor, MUS that employs wells for recharge in dual porosity aquifers faces greater challenges in recovery efficiency because of degraded water quality than systems in more homogeneous aquifers, such as sands and gravels.

METHODS FOR CHARACTERIZATION OF AQUIFERS AND MUS SYSTEMS

To evaluate the conditions for MUS, geoscientists and engineers must determine the aquifer hydrogeologic and hydraulic properties. This section discusses methods to determine these properties, as well as knowledge gained by cycle testing, monitoring, development of a conceptual hydrogeologic framework, and groundwater flow modeling.

Determination of Aquifer Properties Using Laboratory Tests, Pumping Tests, or Slug Tests

Estimating the hydrological properties of water-bearing layers is an essential part of aquifer characterization. Conventional aquifer tests include drawdown (pumping), recovery, interference, and step-drawdown tests (ASCE, 1985). During the test, a well is pumped at a constant rate or stepped rates and

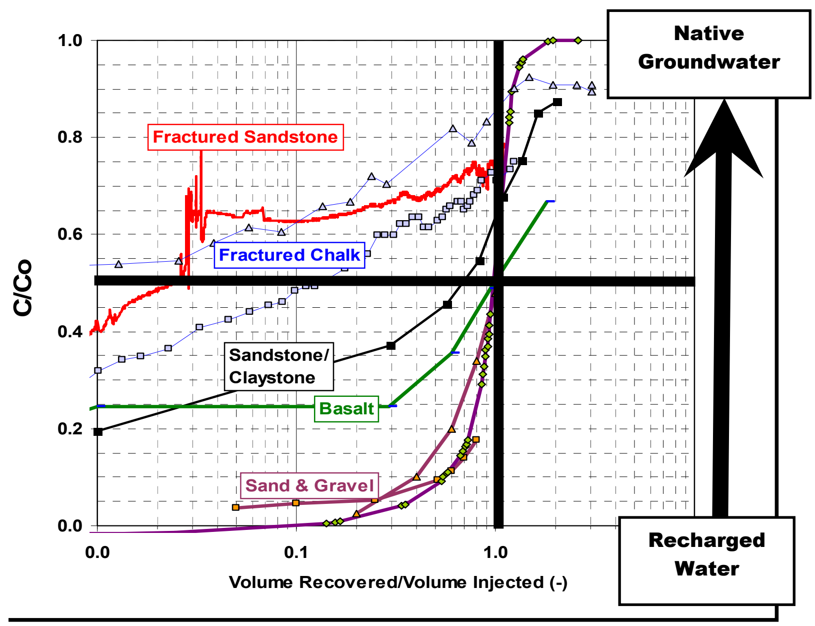

Figure 3-2 Chemical signature of water recovered from ASR wells in various kinds of aquifers, normalized to represent recharged water (conventionally C = 0) and native groundwater (C = 1). Mixture of the two waters is common in high-dispersivity environments (e.g., fractured chalk or sandstone, and to a lesser extent, basalt and heterogeneous sandstone) even at an early time. This suggests that the application of ASR in aquifers with poor water quality faces more challenges in high-dispersivity environments than in lower-dispersivity environments, such as homogeneous sands and gravels. SOURCE: Chris Pitre, Golder Associates, personal communication, March 2006. Reprinted, with permission, from Pitre (2006). Copyright 2006 by Chris Pitre.

variations of water levels with time are observed in the well and/or in one or more observation wells in its vicinity. For a confined aquifer, transmissivity and storativity can be determined by aquifer tests. For an unconfined aquifer, the hydraulic conductivity and specific yield can be determined. For a semiconfined aquifer, the leakance factor and the storativity of the semipervious (leaky) formation can also be determined in addition to the transmissivity and storativity of the aquifer itself (Bear, 1979; Fetter, 2001).

As an alternative to an aquifer test, a slug test or bail-down test can be conducted in a small-diameter monitoring well. In this test, the water level in the well is raised quickly (or lowered), often by lowering (raising) into it a solid piece of pipe (slug). The rate at which the water in the well returns to ambient is measured. Slug tests are used to determine the hydraulic conductivity of the formation in the immediate vicinity of the well (Bouwer and Rice, 1978). A slug test provides an estimate of storativity with low accuracy. Can conventional aquifer tests provide hydraulic parameters needed for evaluation of behavior of the MUS system? To some degree, they can, especially if the duration of the pumping test is long enough for the aquifer response to be representative of a broad area around the well.

However, some limitations exist. First, during pumping, wells tend to remove fine particles from the aquifer, which can improve performance of the well and formation. However, during recharge, wells tend to bring fine particles into the formation, which in turn clogs the formation and well screen. More information on clogging by chemical and biological reactions can be found in Chapters 4 and 6. Second, for an unconfined aquifer, pumping tests cannot characterize the behavior of the vadose zone located immediately above the current groundwater surface, through which the injected water will pass. Therefore, pumping tests cannot be used to replace the injection-pumping cycle test for performance evaluation of injection wells. Third, the heterogeneity and anisotropy of the formation affect the results of the pumping test. Effects of fractured aquifers or carbonate conduits on the pumping tests cannot be identified by a single well pumping test. Therefore, additional pumping tests should be done or a new methodology should be developed to avoid mischaracterization of the formation.

Primary porosity may be determined in laboratory tests from data collected for rock material properties (Moore, 2002).

- Very low primary porosity if the pores are not interconnected or free draining
- Low primary porosity if pores are visible under a l0x hand lens
- Highly permeable if pores are visible to naked eye

Secondary porosity (at least in the form of fractures or solution openings) is not very amenable to laboratory analysis since lab samples are usually too small to include representative amounts of secondary porosity. It is usually inferred by lab analysis from data collected for rock mass properties. Moore (2002) uses the following characteristics to determine the permeability of the material, as based on secondary porosity features: number of joint sets (including the bedding plane), joint aperture, and type of infilling (plastic compared to cohesionless materials). Moreover, the presence of major voids and solution features (caverns, sinkholes, enlarged joints), the occurrence of depositional features (lava tubes or interbedded gravels and lava beds), and the structural setup (faults, stress relief joints) are also indicators of secondary porosity.

A study by Nastev et al. (2004) of hydraulic conductivity measurements obtained by different methods indicates scale-dependent issues. The study, conducted in southwestern Quebec, focused on measured hydraulic conductivity differences between constant-head injection tests, specific capacity tests, pumping tests, and single and multiwell pumping tests. Nastev et al. (2004) demonstrate that at the local scale, groundwater flow is influenced by fractures as opposed to regional flow, which because of the closing of fractures at depth is generally influenced more by the porosity and permeability of the matrix. The availability of fracture zones within an aquifer would increase the potential of preferential pathways allowing higher hydraulic conductivities.

Cycle Injection and Pumping Test and Monitoring for Recovery Efficiency

A cycle injection and pumping test includes recharge of water to be stored and pumping of part or all of the stored water. A comprehensive cycle testing plan can accomplish multiple objectives pertaining to MUS site hydrogeological characterization, operations, and regulatory requirements. For example, tracer tests and analyses of selected physical and chemical parameters can help determine degrees of mixing between source and native groundwater during a cycle test, as well as characterize hydraulic properties of the aquifer. Monitoring and evaluation of injection and pumping pressures can help optimize system performance, identify the presence of dual porosity, and assess reduction in permeability.

Historically, the design of ASR cycle test plans (i.e., recharge volumes and rates, pH adjustment) has focused predominantly on operational issues such as recovery efficiency. Water-quality monitoring for regulatory purposes occurred during cycle testing, but only recently have cycle test plans been designed with more emphasis on scientific issues (e.g., water quality). The same change in emphasis is seen in the decision matrix for placement of monitoring wells. Rather than placing a single monitoring well a standard distance downgradient of the recharge well, more consideration is now given to aquifer anisotropy, adverse hydrogeochemical reactions, and changes in hydraulic gradients throughout all phases of cycle testing. For example, a Wisconsin Department of Natural Resources (WDNR) ASR Technical Advisory Group (2002) states: "The fact that ASR operations will cause frequent changes in directions of groundwater flow in the vicinity of the ASR well means that there may not be a single 'downgradient' direction along which to install a monitoring well. … Thus, strict demonstration of compliance could require numerous monitoring wells… ." With regard to the number of cycle tests needed for a particular ASR well, two primary factors are generally considered: (1) operational testing and optimization, and (2) regulatory requirements. Again, adaptive management effectively guides the decision, while effective communication with regulatory authorities helps streamline the process as detailed in Chapter 6.

In an assessment of the Comprehensive Everglades Restoration Plan (CERP), the National Research Council (NRC) (2001) recognized that limiting factors with regard to recoverability of recharged water include mixing between recharge (source) water and poorer-quality native groundwater and the effects of water-rock interactions. If a cycle test monitoring plan is to fully assess potential water quality changes and system performance that may occur at the scale of full operation, long-term monitoring is required (NRC, 2001), especially if the kinetics of geochemical reactions are not well established.

Although CERP (U.S. Army Corps of Engineers and the South Florida Water Management District, 1999) is orders of magnitude larger in scale and scope than typical ASR operations, most of the issues are the same; therefore similar goals exist for monitoring during cycle testing. These goals can be subdivided relative to which component of a cycle test is involved (i.e., recharge, storage, or recovery) as well as whether the testing should be accomplished during earlier or later cycle tests (U.S. Army Corps of Engineers and the South Florida Water Management District, 2004). Using the project-specific goals of CERP as a foundation, the remainder of this section summarizes what cycle tests can accomplish and what behavior of the aquifer should be monitored:

Before the cycle test, baseline hydraulic and geochemical analyses are conducted on both native groundwater and source water for all parameters to be measured during the cycle test. Borehole geophysical logs completed at this time also provide site hydrogeological characterization, which can be compared to post-cycle test logs to assess potentially adverse changes in borehole characteristics, such as dissolution-related widening of fractures. A standard step-drawdown pumping test is conducted to establish well and formation loss coefficients and well efficiency. Following water level recovery, a long duration pumping test can then be conducted to estimate hydraulic characteristics in the vicinity of the ASR well. Upon completion of the long-term pumping test and associated recovery of water levels to background, a step-injection test is usually conducted to characterize water level response in the ASR well under reverse conditions from the previous step-drawdown test (Pyne, 2005).

During recharge (early cycles), wellhead pressure is monitored and compared to the recharge rate to measure the potential effects of well plugging and estimate the “steady-state” pressure of system. Water samples are collected to evaluate geochemical changes as the source water moves through the aquifer. Tracer tests[1] (tracer added to the ASR well) can be conducted using a monitoring well to guide duration of recharge. There is a need to assess the fate and transport of microorganisms, for which microsphere and/or microphage tracer test can be used. If water quality differences between stored and native water are small and there are no significant concerns regarding geochemical reactions, then a small number of long cycles is appropriate to focus on plugging rates and backflushing frequency required to maintain recharge rates. If there are significant water quality differences between stored and native water, a larger number of cycles is required. After the first cycle, the next three cycles have the same recharge volume and storage period in order to determine the improvement in recovery efficiency with successive identical cycles.

During storage (early cycles), one major concern is the recovery of the stored water. How does the storage time affect recoverability? Samples can be collected from all wells to assess hydrogeochemical reactions with slower rates. If a concern exists with respect to the fate of microorganisms, down-hole diffu-

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[1] These could be completed during any cycle test; with increased cycle testing, preferential pathways will become more developed.

sion chambers may be employed to assess survivability. The storage duration selection process should consider the following factors: anticipated water-rock interactions; the effect of storage on coliform bacteria and other microbiological parameters (if present in the source water), nutrients, metals, radionuclides, and mercury species; and estimated time required for detection of various tracers. If there is a real concern regarding potential geochemical reactions, care should be taken to avoid shocking the formation with a sudden change in quality. Furthermore, storage time should be built into the test program since some reactions such as manganese dissolution require several days or weeks to occur.

During recovery (early cycles), the recovery efficiency can be estimated for comparison to other sites or subsequent cycles. If waters recovered are for the purpose of ecosystem restoration, bioassays may be performed on the recovered water to determine its toxicity to living organisms near the discharge area. This assessment includes the potential effects of mercury bioaccumulation, referring to details in Chapter 4. The water levels or wellhead pressure will be measured along with the pumping rate to assess the effects of water withdrawal on the well and aquifer. Water quality changes in the recovered water and as it moves past monitor wells will be evaluated. Tracer test (tracer added to the monitor well) can be used to provide hydrogeological characterization of the aquifer, and travel time will guide the duration of recovery. The effect of decreased recovery rates on recovery efficiency is also evaluated.

During recharge (later cycles), besides monitoring the same parameters as in early cycles, more parameters for long-term performance of the MUS system are collected. The system will be operated based on projected conditions envisioned for an anticipated larger-scale system. The subsurface storage volume will be built up. The effect of buoyancy on system efficiency is evaluated using longer recharge duration. Upward migration of recharge water will also be evaluated by monitoring units above the confining units.

During storage (later cycles), the system will be operated based on projected conditions at full scale. The effect of buoyancy on system efficiency is evaluated using longer storage duration.

During recovery (later cycles), the system will be operated based on projected conditions at full scale. The potential for upconing due to longer periods of recovery is assessed. The water quality changes will also be assessed to better define recovery of the stored water.

If multiple ASR and monitoring wells are being tested, the characteristics of the storage volume (shape, thickness, expansion rate, etc.) can also be assessed. The maximum pressure buildup ASR well field can be evaluated. Post-cycle test borehole logging can be used to assess physical changes in an aquifer due to repeated cycle testing.

Hydrogeological Framework of an MUS System

Geologic and geophysical data collection from airborne surveys, land surface, and boreholes, coupled with hydrologic data from various tests at the laboratory and field scale, comprise the foundation of knowledge required to develop a robust conceptual hydrogeological framework. This framework should be developed at multiple scales to accommodate site-specific MUS system planning needs as well as regional water supply needs. Surface geologic maps, cross sections, and subsurface maps (e.g., surfaces and thicknesses of underground units) of lithostratigraphic and hydrostratigraphic units (including aquifers and relative confining units) characterize the framework. Integration of geologic, hydrogeologic, and hydraulic data with the hydrogeological framework facilitates modeling and assessment in support of MUS.

Three-Dimensional Models for Aquifer Characterization

The amount and types of data required for aquifer characterization depend on the heterogeneity of the aquifer and the optimum resolution required to develop models on which the MUS system design will be based. Such data may originate from multiple sources (i.e., consultants; local, state, and federal agencies) and multiple projects. Amassing multiple data types from disparate sources comes with challenges, including quality assurance and quality control (QA/QC), data standardization (e.g., units, methodologies), metadata, and resolution issues. While development of such databases requires significant financial and human resources, organization of these data into a seamless application facilitates effective use of time and funds over the duration of long-term MUS projects.

Continuing advancements in geographic information systems (GIS) and hydrologic computer models facilitate integration of complex databases with three-dimensional (3D) applications. Storage of geologic or hydrogeologic data in three dimensions allows interpolation of 3D hydrogeologic units, designation of measured or interpreted properties to the units, volume calculations, morphology analysis, representation of complex fault systems, parameter flux (i.e., groundwater flow, chemical diffusion) between units, and interpolation of hydrologic properties within the unit volumes. Once the 3D framework is established, "virtual" cross sections (e.g., geologic, hydrologic, geophysical, hydrochemical) and borehole stratigraphy can be predicted and represented graphically. Moreover, 3D visualization of aquifer properties, which is available through commercial software packages, can be rendered.

Ross et al. (2005) state that "3D geomodeling is expected to become a standard in the near future." Numerous advantages exist regarding integration of complex database solutions with 3D hydrogeologic framework mapping (e.g., Artimo et al., 2003; Faunt et al., 2005; Ross et al., 2005; Soller et al., 1998; Thorleifson et al., 2005): (1) the 3D framework model is comprised of relational discrete surfaces or volumes representing best-available data in a common and internally consistent framework that does not require high-end computer power; (2) the degree of complexity of the framework can be modified to fit the needs of individual projects and the development of derivative products (e.g., flow simulations, cross sections, aquifer vulnerability mapping);(3) the 3D framework model not only facilitates development of groundwater flow models, but also provides a feedback loop (adaptive modeling) between the framework and groundwater model, allowing for refinement of both models to reduce uncertainty; (4) data redundancy is minimized and data standardization is maximized; (5) data gaps can be efficiently identified, such as optimizing the location, construction, and depth of wells to address specific data needs; (6) automated data entry and semiautomated QA/QC can be streamlined (7) this facilitates GIS and data analyses using complex data sets, including graphic logs; (8) once implemented, the time required for data entry and model development is significantly reduced; and (9) 3D visualization (static or animated) helps scientists, engineers, and environmental managers more fully understand the dynamics and complexities of the hydrogeologic system.

These advantages, however, are accompanied by caveats, especially with regard to spatial resolution and interpolation uncertainty. For example, the correlation length of hydraulic conductivity measured in a core sample may be on the order of a few meters. In such a case, interpolative maps drawn by GIS, 3D mapping applications, or other methods using kriging or similar algorithms are to be interpreted with care if the data set is separated by more than this distance. Potential misuse of 3D maps or models is minimized through assessments of interpolated surfaces or volumes that consider factors such as map prediction error and effects of data quality, gaps, and clusters. Box 3-4 shows a case study of 3D geologic modeling and database solutions in an MUS system.

Additional MUS applications of aquifer framework characterization include development of models or tools that identify (1) cultural impediments to MUS, (2) hydrogeologic settings and land use applications that increase the contamination potential of an MUS storage zone, and (3) storage zones suitable for specific MUS activities.

Examples of these tools include ASR suitability scoring (Brown, 2005) or ASR potential mapping (Dudding et al., 2006), and aquifer vulnerability assessments (e.g., Doerfliger et al., 1999; Huaming and Wang, 2004; Arthur et al., 2007).

Tracers, Geophysics, and Other Aquifer Characterization Methods

Tracers. In MUS systems, there are four general applications for tracers: (1) assess the fate and transport of microorganisms; (2) determine aquifer properties such as porosity and hydraulic conductivity; (3) determine movement of the recharge water, including the degree of mixing between recharge and native waters, as well as dispersion and diffusion; and (4) evaluate in situ reaction rates

BOX 3-4
Case Study: The Role of 3D Geologic Modeling and Database Solutions in the Virttaankangas Aquifer Artificial Recharge Project, Southwestern Finland

The purpose of the Virttaankangas Aquifer artificial recharge project is to provide the 285,000 inhabitants of the Turku area, southwestern Finland, with good-quality potable water by 2010. The total budget of this project will be about 100 million euros. Pretreated river water from the Kokemäenjoki River will be conducted by pipeline to the Virttaankangas Aquifer for infiltration. To provide acceptable water quality, the residence time of the water in the aquifer is designed to be at least three months. The water will then be pumped from the aquifer to the Turku region for consumption.

The costs of planning and building the 100-km pipeline and associated infrastructure are the largest items of expenditure in the project. However, research on the geology and hydrogeology of the Virttaankangas Aquifer will be critical to its success.

In this project, geological, geochemical, and geophysical data are organized within an integrated database solution that also accepts manually and automatically measured groundwater field data. Among the wide array of data types are sedimentological, ground-penetrating radar, gravimetric, isotopic, and physical water quality parameters; results of pump and infiltration tests; and hydraulic heads. Through implementation of this database, all hydrologic and hydrogeologic data are accessible "on demand" for development of semiautomated, internally consistent 3D model units, from which hydrogeologic framework models, and subsequently, groundwater flow models are generated. This dynamic and flexible database will be used and expanded through the development and production phase of this MUS project.

SOURCE: Artimo et al. (2005).

(see Chapter 4). Tracer studies are also needed to address regulatory compliance and hydrogeological and hydrogeochemical characterization. Some states, for example, require reporting of residence times and travel distances for recharge water (Shrier, 2002), as well as demonstration of microbial die-off (EPA, 2006). Dispersivity and the degree of mixing can be the single most important factors in recovery efficiency, which is often a measure of MUS system performance (Brown, 2005).

Chemical tracers allow distinction between the two waters of interest. These tracers include basic water quality parameters, stable and radiogenic isotopes, and constituents added to the recharge water to give it a unique "signature." To provide optimal results, the tracer should have the following characteristics:

- impart or represent a unique physical or chemical characteristic of the recharge water,
- be nonreactive with the aquifer matrix,
- be nonreactive with any combination of the two waters,
- be photochemically stable,
- be unaffected by microbial activity,
- be easily measured, and
- not be readily sorbed.

For example, Cl^- can serve as a natural conservative tracer if no evaporite minerals are present in the aquifer and if source water concentrations are consistent (i.e., no evaporative concentration of Cl^- occurs within reservoirs of recharge water). In a similar manner, F^- is generally a useful tracer as long as fluorine-bearing minerals are not present in the aquifer or predicted to precipitate based on water-rock geochemical (equilibrium) models. Br^- is another tracer in the same family. Visible fluorescent dyes such as rhodamine WT and fluorescein, which are not sorbed by the aquifer materials, can also be useful; their presence can be detected visually or with the use of fluorometers or ultraviolet light. Organic contaminants such as ethylenediaminetetraacetic acid (EDTA), which may be present within recharge basins, can also serve as indicators of the recharge water. Additional water quality parameters commonly used as tracers include, but are not limited to, NO_3^-, SO_4^{2-}, and boron. Under certain conditions, including a sufficient contrast between the two waters, parameters such as electrical conductivity, total dissolved solids (TDS) and even temperature can be used as tracers, at least to a qualitative degree. Stable isotopes of the water molecule itself (e.g., ^{18}O, ^{2}H [deuterium]) can be used to determine ratio of mixing when waters of different origins are mixed.

"Emerging" chemical tracers may also be used. The rare-earth element gadolinium (Gd) is an example. Magnetic resonance imaging (MRI) technology uses a gadolinium-based acid to improve contrast of the image. This stable acid compound is passed through the human body into wastewater and is not removed during effluent treatment. Knappe et al. (2005) found that the gadolinium compound has many of the characteristics outlined above, and as a result, it was a useful tracer in their study of bank filtration. Additional emerging tracers include endocrine disrupting compounds (EDCs) (e.g., Verstraeten et al., 2005).

The use of *microbial tracers* for groundwater movement has been around since the late nineteenth century (Harvey and Ryan, 2004). However, biological tracers have been used extensively in studying groundwater movement only since the 1970s (Wimpenny et al., 1972).

Viruses were first used to study the hydrology of aquifers in the early 1970s (Martin and Thomas, 1974). The viruses of choice were bacteriophages; these viruses infect bacterial hosts. Under certain circumstances, bacteriophages are preferred over colored dyes as tracers for water movement because they can easily be obtained in high titer in a relatively small volume (e.g. 10^{14} pfu in 1-10 L versus 30-40 L for Rhodamine WT dye). In addition, because of the host specificity of the phages, different bacteriophages can be injected at the same time or at different time in the same water system (Rossi et al., 1998). Biological tracers (in this case, bacteriophages) may represent a better model for the transport behavior of waterborne pathogens in water because they are present as colloids whereas chemical tracers are generally water soluble. Bacteriophages that have been used extensively in hydrology studies as surrogates for viral pathogens are PRD1(P22) and MS2 (Yahya et al., 1993).

Salmonella typhimurium, strain LT-2, is one of the most commonly used hosts for propagation and enumeration of P22 in environmental studies (Harvey and Ryan, 2004). Because of its small size and double-stranded DNA nature, P22 has been found to be very stable in natural environments compared to other viruses and can be used as the worst-case scenario model. An inactivation rate of 0.2 log per day in river water was observed at 21-25°C (John and Rose, 2005b).

Overall, tracer tests can be highly useful, but they do have limitations. The expense may limit the tests to a few available monitoring wells whose location and spacing may not have been designed with tracer tests in mind. Also, during recovery tests, regulatory limitations may pose a severe constraint on tracer recovery due to other parameters (e.g., chloride, TDS) that must be disposed of or discharged into nearby waterways. These constraints may affect pumping tests as well.

Hydrogeophysical Methods. Hydrogeophysical technology developed in recent years provides qualitative and quantitative information about subsurface hydrological parameters or processes (Hubbard and Rubin, 2005). Various platforms can be used to collect data to characterize hydrological parameters and processes at different stages of testing and operation of an MUS system. Hydrogeophysical surveys for an MUS system can range from laboratory (or point) scales (10^{-4} to 1 m), local scales (10^{-1} to 102 m), to regional scales (101 to 105 m). Several factors are considered when selecting a characterization or data acquisition approach for MUS system applications: the objective of the investigation relative to the sensitivity of different geophysical methods; the desired level of resolution; site conditions (e.g., power lines or other cultural impediments); available time, funds, and computational resources; experience of the investigator; and availability of geologic or hydrologic data for calibration or verification of the geophysical data.

The most common land surface or airborne (fly-over) hydrogeophysical studies are electromagnetic (EM) induction (both frequency and time domain), gamma-ray spectrometry (radiometrics), and magnetics (Paine and Minty, 2005). Electromagnetic induction, including both frequency-domain EM (FDEM) and time-domain EM (TDEM) approaches, measures the apparent electrical conductivity of the ground to depths ranging from a few meters to a few hundred meters, depending on the instrument resolution and the ground conductivity (Everett and Meju, 2005). Bulk conductivity of the ground is a function of water content, water chemistry, pore volume and structure, and electrical properties of the host mineral grains (McNeill, 1980). Gamma-ray surveys map the distribution of radioactive elements—potassium (K), uranium (U), and thorium (Th), at the earth's surface, which varies with source rock mineralogy and surface processes such as erosion, pedogenesis, and sediment deposition as well as human activities (development of an MUS system).

Surface geophysical methods such as seismic refraction and reflection (Pride, 2005), microgravity, controlled source audiofrequency magnetotelluric (CSMAT) profiling, TDEM, and resistivity surveys may help identify heterogeneities in permeability, including preferential flow paths. For example, Dobecki et al. (2007) employed CSMAT before, during, and after an ASR recharge event to characterize the distribution of recharge water within the storage zone. Some of these methods require certain site characteristics including an order-of-magnitude resistivity contrast between recharged source water and native groundwater. Target depth and cultural noise (e.g., pipelines, large grounded metal structures) also affect the ability of EM methods to resolve features. Related emerging technologies involving an induced magnetic field via groundwater low-voltage charging (Rollins, 2006) are also of potential application for MUS.

Ground-penetrating radar (GPR) can be employed to address numerous hydrogeological questions, ranging from geological structure to material properties (Annan, 2005). It can delineate fine-scale depositional stratigraphy and its spatial variations, which are important in terms of understanding correlation lengths and the scale of heterogeneity for hydraulic conductivity (Annan, 2005; Bristow and Jol, 2002; van Overmeeren, 1998). GPR sensitivity to water content also provides a technique for mapping groundwater surfaces including perched water tables.

During 2005, the U.S. Geological Survey (USGS) continued its work on a web-enabled earth resistivity tomography (ERT) monitoring system that will be used to assess and monitor hydrologic processes including ASR and saltwater intrusion into coastal aquifers. Monitoring in a dual-porosity system can be very problematic. Due to the high ratio of intragranular porosity to open spaces in the aquifer (conduits, fissures, fractures, etc.), storage zone monitoring wells will likely not represent the chemical and physical conditions in the open storage zone network. Moreover, if geologic processes that led to the development of dual porosity have affected zones above or below the storage zone, confinement of the storage zone, if needed (as in ASR), is compromised and recovery efficiency will decline.

Borehole geophysics includes all methods for making continuous profiles or point measurements at discrete depths in a borehole using different types of probes (Kobr et al., 2005). One of the most important attributes of geophysical log tools is the ability to make several different physical or chemical measurements in a borehole. Examples include spontaneous potential; normal and/or lateral resistivity logs; conductively focused current logs and micro-focused logs; gamma-ray logs; gamma-gamma logs; neutron logs; elastic wave propagation logs; acoustic televiewer; and temperature logs. These methods can be employed to delineate stratigraphy; determine bulk density, porosity, and moisture; and characterize the structure of aquifer materials (orientation of fractures, fracture openings and bedding), water movement (vertical and horizontal), and water quality. Each method has its own limitations; therefore, multiple methods are employed to have a better understanding of the aquifer system.

Dual or secondary porosity in an MUS storage zone can be identified using caliper logs, which provide borehole diameter. Poorly consolidated materials, washout zones, and possible fractures or conduits are among the features that can be identified. Loss of drilling fluids or circulation may indicate influence of secondary porosity as well. Borehole video logs also help identify fractures, cavities, and conduits intersecting the borehole. These video logs, as well as borehole flowmeters, allow measurement of groundwater flow rates and directions. These methods may be complemented by microgravity surveys to detect conduits.

Cross-borehole ERT is a method that applies what has historically been a surface geophysical survey, EM. Cross-borehole ERT provides a vertical profile of resistivity (e.g., a resistivity cross section) between boreholes. Results of multiple cross-borehole ERT surveys can be combined to construct 3D distributions of groundwater quality, which can be validated by hydrochemical sampling. In a 2D application, Johnson et al. (2004) designed an ERT survey to monitor the injection of relatively low-resistivity water into a brackish-water fractured limestone aquifer utilized as an ASR storage zone.

Table 3-3 lists several common geophysical characterization methods, which are classified according to their acquisition category (Hubbard and Rubin, 2005). The attribute that is typically obtained from each method is given, along with some examples of hydrogeological objectives for which each method is particularly well suited. These objectives can be broadly categorized into three key areas: hydrogeological mapping, hydrogeological parameter estimation, and monitoring of hydrological processes.

Modeling of Groundwater Flow During Recharge, Storage, and Recovery

A groundwater model is a simplification of a real-world aquifer system that provides a cost-effective instrument for planning, design, or operation of MUS. Models are used largely to understand the behavior of a flow system and to predict how the system will behave in the future (Fetter, 2001), they can be built as analytical or numerical models. Analytical models are derived from differential equations that describe the distribution of hydraulic heads in space (x, y, z) and time. Storage properties and hydraulic conductivities in a groundwater system can sometimes be solved analytically when simplified. Among those assumptions may be homogeneity, flow exclusively in one or two dimensions, and simple boundary conditions (e.g., time and space invariant). Therefore, analytical models are elegant and useful in wellhead protection, and dewatering, among other studies. However, they are less versatile than numerical solutions since the problem has often been simplified.

Numerical models simulate groundwater flow using algebraic equations. Heterogeneity, three-dimensional flow, and complex boundary conditions are more easily incorporated into numerical equations. These models require more data, conceptualization, design, and expertise. Equations can be solved via finite differences or finite elements, which are among the most used methods. Most often, a computer code solves the flow equations by applying approximation techniques. In selecting the type of model for use, it is necessary to determine whether the model equations account for the key processes occurring at the site. Each model, whether it is a simple analytical model or a complex numerical

TABLE 3-3 Common Geophysical Characterization Methods that are Used to Assist in Hydrogeological Investigations

Acquisition Approaches	Characterization Methods	Attributes Typically Obtained	Examples of Hydrogeological Objectives
Airborne	Remote Sensing	Electrical resistivity, gamma radiation, magnetic and gravitational field, thermal radiation, electromagnetic reflectivity	Mapping of bedrock, freshwater-saltwater interfaces and faults, assessment of regional water quality
Surface	Seismic refraction	*P*-wave velocity	Mapping of top of bedrock, groundwater surface, and faults
	Seismic reflection	*P*-wave reflectivity and velocity	Mapping of stratigraphy, top of bedrock, and delineation of faults or fracture zones
	Electrical resistivity	Electrical resistivity	Mapping aquifer zonation, groundwater surface, top of bedrock, freshwater-saltwater interfaces and plume boundaries estimation of hydraulic anisotropy, and estimation or monitoring of water content and quality
	Electromagnetic	Electrical resistivity	Mapping aquifer zonation, groundwater surface, freshwater saltwater interfaces, and estimation or monitoring of water content and quality
	Ground-penetrating radar	Dielectric constant values and dielectric contrasts	Mapping of stratigraphy and groundwater surface, estimation and monitoring of water content
Crosshole	Seismic	*P*-wave velocity	Estimation of lithology and fracture zone detection
	Electrical resistivity	Electrical resistivity	Mapping aquifer zonation and estimation or monitoring of water content and quality
	Radar	Dielectric constant	Estimation or monitoring of water content and quality, mapping aquifer zonation
Wellbore	Geophysical well log	Electrical resistivity, seismic velocity, and gamma activity	Lithology, water content, water quality, and fracture imaging
Laboratory/ Point	Electrical, seismic, dielectric, and x-ray methods	Electrical resistivity, seismic velocity and attenuation, dielectric constant, and x-ray attenuation	Development of petrophysical relationships, model validation, investigation of processes and instrumentation sensitivity

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model, may have utility in hydrogeological and remedial investigations. The key is to make certain that the problem is clearly defined and that the selected model is the best choice to answer the posed questions.

Analyses using groundwater and solute transport numerical modeling may help to

- Evaluate the performance of a regional set of ASR wells to aid in establishing the design, spacing, orientation, and capacity of those wells;
- Evaluate regional changes in hydraulic head and flow patterns;
- Evaluate the impact on the environment, including neighboring surface water flow, and existing users;
- Evaluate the critical pressure for rock fracturing or widening of existing fractures;
- Analyze the relationship between storage interval recovery rates and recharge volume so that the recovery of water is optimized;
- Visualize the movement of stored water throughout the wellfield, which is of special interest where the storage zone contains water of lesser quality or where dual porosity is present; and
- Evaluate the extent of potential water quality changes in the aquifer during storage and movement.

Modeling Protocol

A protocol should be followed when developing a model, as documented in relevant ASTM (American Society for Testing and Materials) guidelines and other literature, such as Anderson and Woessner (1992), Spitz and Moreno (1996), and Mercer and Faust (1981). There are several important steps to take during model development.

First, the purpose of the model should be identified clearly, and it should be determined whether modeling is the appropriate type of analysis. The next step requires building the conceptual model, which can be a pictorial and/or a written description of the real-world aquifer system and the simplifying assumptions that describe the primary hydrogeologic processes. The process of building a conceptual model starts by identifying the boundaries of the model—any physical (e.g., faults) and/or hydraulic (e.g., groundwater divide, large water body, ocean) boundaries within the area of interest, formulating the general water budget (evapotranspiration, baseflow, pumping, etc.), and defining the flow system from water levels or contour maps in terms of the locations of recharge and discharge areas and the connection between surface and groundwater systems. Field data are collected for hydraulic properties of the aquifer (e.g., hydraulic conductivity, storage). Estimates of the vertical aquifer properties are especially important for an MUS model, since vertical flow through an aquifer and/or confining unit can play a large role in determining flow behavior when the system is stressed hydraulically (i.e., ASR recharge or recovery). Typically, building the conceptual model is an iterative process, which requires reevaluating the conceptual model throughout the model development process.

The steps described in the conceptual model must be formulated mathematically using appropriate governing equations. In the case of MUS, governing equations for groundwater flow, variable density (if this is the case) and transport should be solved accurately by the numerical method applied in the selected computer code. Standard flow models assume that the density of groundwater is constant, which is reasonable if salinity and temperature show little temporal or spatial variance. However, differences in the density of miscible fluids, associated with saltwater intrusion, or freshwater recharge into brine aquifers, among other processes, require the use of models that solve for solute transport and/or temperature in addition to flow (Brown et al., 2006). It should be noted that solving for groundwater flow with variable density and transport may require large run times. If the model run time becomes unmanageable, it may be necessary to re-visit the model design (e.g., using a coarser discretization or aggregating the stress periods).

The next step is MUS model design, which sets the spatial and temporal discretization of the grid. A refined grid improves model stability, and accuracy may be improved while numerical dispersion in the solute transport components is reduced although this may have a negative effect on the model run time. A difficult modeling problem exists when a combination of steep head gradients and sharp concentration fronts is present and could result in model instability and inaccuracy. Brown et al. (2006) found that the concentration changes around an ASR well are focused within 250 feet of the well (pumped at a rate of 5 Mgal/d) when simulating ASR in the Floridan Aquifer System; therefore, a constant horizontal grid or mesh resolution within that radius can be established with an increase in grid size at a reasonable ratio beyond 250 feet. ASR wells with large recharge or recovery rates may need additional horizontal and vertical grid refinement (Brown et al., 2006). Excessive refinement is not advised so as to maintain reasonable run times for calibration and prediction. In effect, finer resolution should be employed at the areas of interest, while areas of less interest may be modeled with a coarser resolution.

Meeting specified calibration criteria or targets by reproducing measured water levels or flow rates is the goal of model calibration. The MUS model builds on calibration against seasonal water levels and water quality, as well as performing transient calibration at existing ASR and well sites where aquifer test or ASR cycle testing data are available. Once the model is calibrated, a sensitivity analysis is recommended to quantify and show the effects of uncertainty in the calibrated model.

Modeling Software

Modeling an MUS project can be a daunting challenge. Fortunately, there are several, available model codes for both finite-element and finite-difference methods that could be used for MUS projects. The several models are available in both the public and the private domain. A description of the model codes, which are available in the public domain, is presented herein including: MODFLOW-MT3D, SEAWAT, HST3D, SUTRA, and WASH123D. Several other established codes could also have been selected, including FEFLOW (a proprietary code from Europe that may be more difficult to procure for U.S. government work efforts; Pavelic et al., 2004, 2006). Each code exhibits both strengths and weaknesses. All of the codes provide much of the model functionality desired for the MUS Regional Study for saturated flow. WASH123 codes provide the best overall functionality for any type of MUS due to the coupling of surface and groundwater components.

MODFLOW is a three-dimensional finite-difference groundwater model that was first published by the USGS in 1984 (McDonald and Harbaugh, 1988) and has been updated periodically. The groundwater flow equation is solved using the finite-difference approximation. The flow region is subdivided into a grid; within each cell, properties are assumed uniform. The model layers can have varying thickness. A flow equation is written for each cell. Several solvers are provided for solving the resulting matrix problem. MODFLOW is considered to be the most widely used program for constant-density groundwater flow problems. Key factors for MODFLOW's popularity in the modeling community are its thorough documentation, its modular structure, and the public availability of the software and source code. The major limitations of MODFLOW are that the model cannot provide a water budget for the full hydrologic cycle because overland flow and the unsaturated zone are not simulated. However, MODFLOW 2005 has a new package with the capability to simulate unsaturated flow.

MODFLOW-MT3DMS is a suitable tool for a mass-balance approach to evaluating storage and recovery, but is not suitable if there are significant density issues in the study area. MT3DMS (Zheng and Wang, 1999) is based on a modular structure to permit simulation of solute transport and particle tracking (dispersion or advection). MT3DMS interfaces directly with MODFLOW. MT3DMS is a 3-D transport model, where MS denotes the multispecies structure for accommodating add-on reaction packages. MT3DMS has a comprehensive set of options and capabilities for simulating advection, dispersion or diffusion, and chemical reactions of contaminants in groundwater flow systems under general hydrogeological conditions (Zheng and Wang, 1999).

SEAWAT (Guo and Langevin, 2002) is a computer program for the simulation of three-dimensional, variable-density, transient groundwater flow with solute transport in porous media. The program combines MODFLOW and MT3DMS (Zheng and Wang, 1999) into a single computer program that solves the coupled flow and solute transport equations. A disadvantage in using this code is the long model run times due to small time-step requirements for contaminant transport models.

HST3D (Kipp, 1997) is a heat and solute transport program that simulates groundwater flow and related heat and solute transport in three dimensions. This program may be used to analyze subsurface waste injection, saltwater intrusion, and freshwater recharge and recovery (Kipp, 1997). This code solves three-dimensional, saturated groundwater flow with heat and solute transport. These three equations are coupled through the dependence of advective transport on the interstitial fluid velocity field, the dependence of fluid viscosity on temperature and solute concentration, and the dependence of fluid density on pressure, temperature, and solute concentration.

SUTRA (Voss and Provost, 2002) is a model for saturated-unsaturated, variable-density groundwater flow with solute or energy transport. SUTRA (saturated-unsaturated transport) can simulate fluid movement and transport of either energy or dissolved substances in a subsurface environment. The code employs a two- or three-dimensional finite-element and finite-difference method to approximate the governing equations that describe the two interdependent processes that are simulated: (1) fluid density-dependent saturated or unsaturated groundwater flow; and (2) either transport of a solute in the groundwater or transport of thermal energy in the ground water and solid matrix of the aquifer. SUTRA energy transport simulation may be employed to model thermal regimes in aquifers, subsurface heat conduction, aquifer thermal energy storage systems, geothermal reservoirs, thermal pollution of aquifers, and natural hydrogeologic convection systems. SUTRA has been used for past ASR simulation studies. Voss (1999) provides a review of SUTRA applications.

WASH123D (Watershed Systems of 1D Stream-River Network, 2D Overland Regime, and 3D Subsurface Media) is a public domain model developed by the Waterways Experiment Station for the U.S. Environmental Protection Agency (Yeh, 1998). The EPA and the U.S. Army Corps of Engineers endorse WASH123D for modeling of comprehensive watershed management plans. WASH123D is a finite-element numerical model designed to simulate variably saturated, variable-density water flow and reactive chemical and sediment transport in watershed systems. It is capable of representing a watershed system as a combination of 1D river or stream, 2D overland, and 3D subsurface subdomains. WASH123D is a physically based, spatially distributed, finite-element, integrated surface water and groundwater model. WASH123D is applicable to a variety of problems, including flood control, water supply, water quality, structures, weirs, gates, junctions, evapotranspiration, and sediment transport for both event and continuous simulations. WASH123D can provide a water budget for the full hydrologic cycle.

The groundwater flow portion of the code utilizes an adaptation of the FEMWATER code (Lin et al., 1997). A disadvantage of using this code is the long model run times due to small time-step requirements for contaminant transport models. WASH123D has been applied in south Florida (Brown et al., 2006).

Data Needs

Data needs can be extensive for modeling MUS projects. Developing a groundwater flow model is always the first step before adding the variable-density, solute transport, and/or heat components. As discussed in the modeling protocol, it is important to define the hydrogeologic system in terms of aquifer characterization and hydraulic properties. The physical framework is defined by the characteristics of the hydrostratigraphic layers and boundary conditions. Geological maps and cross sections should be shown and should identify the hydrostratigraphic units. Standard groundwater flow model properties, such as hydraulic conductivity or transmissivity and storativity, must be defined spatially and in the context of the physical framework. In addition, a topographic map, of basins and surface water bodies should be compiled. The extent and thickness of stream and lake sediments are important to show the connection between surface and groundwater systems.

Box 3-5 describes a case study of applications of WASH123D in an MUS system in a brackish aquifer in south Florida.

Other data needs include properties that describe water quality. These properties are of special importance to MUS projects, especially when waters of different qualities may be combined. In MUS, the ambient (native) water quality refers to water measured upstream (or outside) of the influence of a pollutant or contaminant during average flow conditions. Water that is no longer fit for use is said to be polluted or contaminated but legal standards change according to use, such as environmental, human consumption, irrigation, or others.

One measure of water quality is total dissolved solids, which is defined by the weight of the solids that remain after the water sample is completely evaporated (milligram per liter). Among the major constituents (natural constituents) are calcium, magnesium, sodium and potassium, chlorine, sulfite, carbonate, bicarbonate, and silicon. Minor constituents are iron, manganese, fluorine, nitrate, strontium, and boron. TDS can also be classified as fresh, brackish, saline, and brine (Fetter, 2001). Other quality standards are associated with dissolved oxygen, the presence of radioactive constituents, and bacterial content (see Chapter 4 for details).

Regional- Versus Local-Scale Modeling

Models can be applied to address both regional- and local-scale issues. Model applications can assess system-wide impacts as well as other impacts from the local, complex geometry (e.g., dual porosity, changes in the storage zone) of the MUS System. Models of different spatial, and sometimes temporal, scales are needed to address regional and local issues.

Regional models are tools used for evaluating the cumulative effects of multiple, and potentially competing, uses of water resources in a region. These models are valuable tools for (1) determining regional changes in aquifer heads

BOX 3-5
Case Study: ASR Modeling for a Brackish Aquifer in South Florida

Mixing of fresh water and native water from a brackish aquifer can be modeled with the WASH123D numerical model, which computes solute transport and density-dependent flow. A hypothetical case for the Comprehensive Everglades Restoration Plan is presented to show a typical model design and to depict results from an ASR site. The WASH123D finite-element code is applied to simulate injection, storage and recovery using a box model with a domain of approximately 40 miles x 40 miles x 2,340 feet.

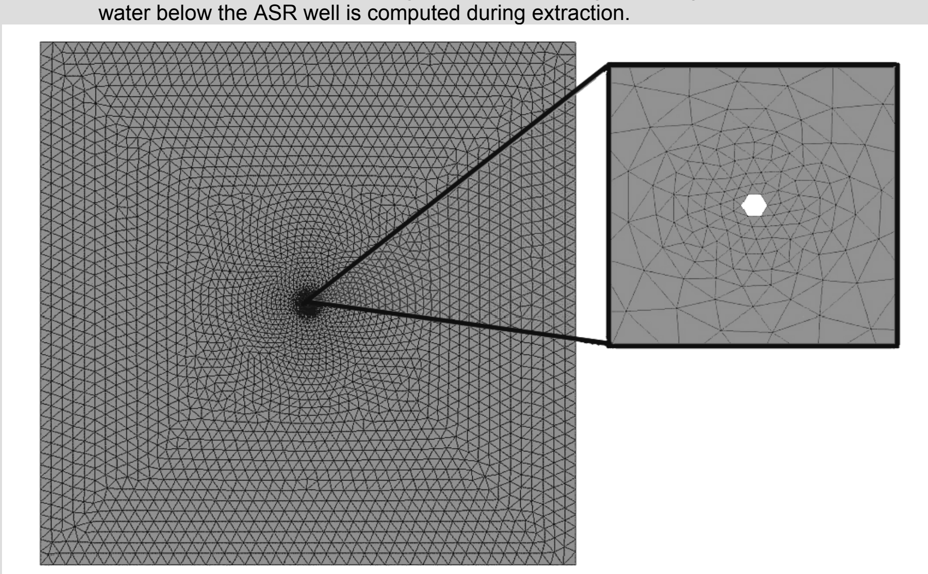
Flow and concentration gradients may be high in the neighborhood of an ASR well. Refinement of the vertical and horizontal resolution of the 3D mesh in the vicinity of the ASR well may be necessary. In the hypothetical case, the horizontal mesh resolution at the ASR well is 10 feet and expands gradually to 5,000 feet along the model perimeter as illustrated in part A of the figure below. Vertical mesh resolution, described in part B, varies among the different conceptual geologic units, which represent the Surficial Aquifer System (SAS), Hawthorn Group (HG) confining unit, and Florida Aquifer System (FAS) including the Upper (UFA), Middle (MF) and Lower Floridan (LF). Vertical resolution is increased in the confining units directly above and below the recharge zone (i.e., UFA). This increased resolution allows the model to depict the large head and concentration gradient at the interfaces of these confining units.

The results of a sensitivity analysis indicate that the change of the computational result becomes insignificant as the time step size is reduced to less than 0.5 day. Therefore, a time-step size of 0.5 day is used. The boundary conditions are applied to the element faces representing the well screen within the UFA. Constant boundary conditions were used to assign the total head along the eastern and western model boundaries. No-flow boundary conditions were used along the northern and southern model boundaries. Constant boundary conditions were also used to assign the concentration along the model perimeter.

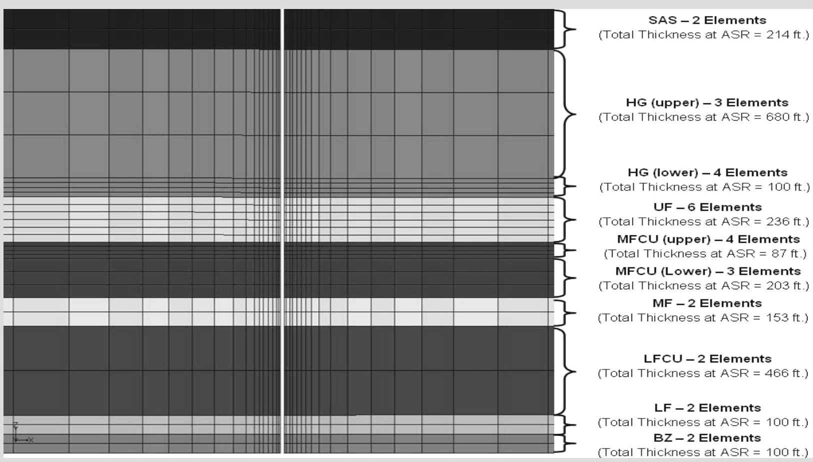
The model simulated a 30-day recharge period, followed by a 305-day storage period and a 30-day recovery period. The following sections discuss the results of the WASH123D model during each simulation phase.

- *Initial Condition*. The nodes in the geologic units above the UFA were assigned a constant concentration of fresh water (150 mg/L), while the nodes in and below the UFA were assigned a constant concentration of seawater (35,000 mg/L). Refer to part B of the figure for an understanding of the assignment of the initial conditions and location of ASR well prior to recharge.
- *Recharge Phase*. Starting with the initial conditions, the ASR pumping well injects fresh water (with a concentration of 150 mg/L) into the UFA at a rate of 5 Mgal/d for 30 days. The hydraulic head at and immediately surrounding the ASR well increases nearly doubling in magnitude. Part A of the figure shows a cross-sectional view of the concentration profile in the vicinity of the ASR well at the end of the injection cycle, it shows that the injected fresh water has displaced the ambient saline water forming a “spheroid” of lower concentration water in the vicinity of the ASR well.
- *Storage Phase*. After the recharge phase, the ASR well is turned off for 305 days. During this storage period, the hydraulic conditions stabilize close to steady-state conditions. Part B of the figure shows a cross-sectional view of the concentration profile in the vicinity of the ASR well at the end of the storage period. Although the concentration at the ASR well remains relatively constant, the effects of buoyancy stratification are noticeable. During the storage period, the density effect is the dominant factor in the flow fields. The concentrations at the lower portion of the UFA increase substantially faster than at the top of the aquifer.

- *Extraction Phase*. After the storage period, the ASR recovery begins at a rate of 5 Mgal/d for 30 days. During this extraction cycle the hydraulic head at and immediately surrounding the ASR well decreases substantially. Part C of the figure shows a cross-sectional view of the concentration profile in the vicinity of the ASR well at the end of the storage period. Up-coning of the higher-concentration water below the ASR well is computed during extraction.



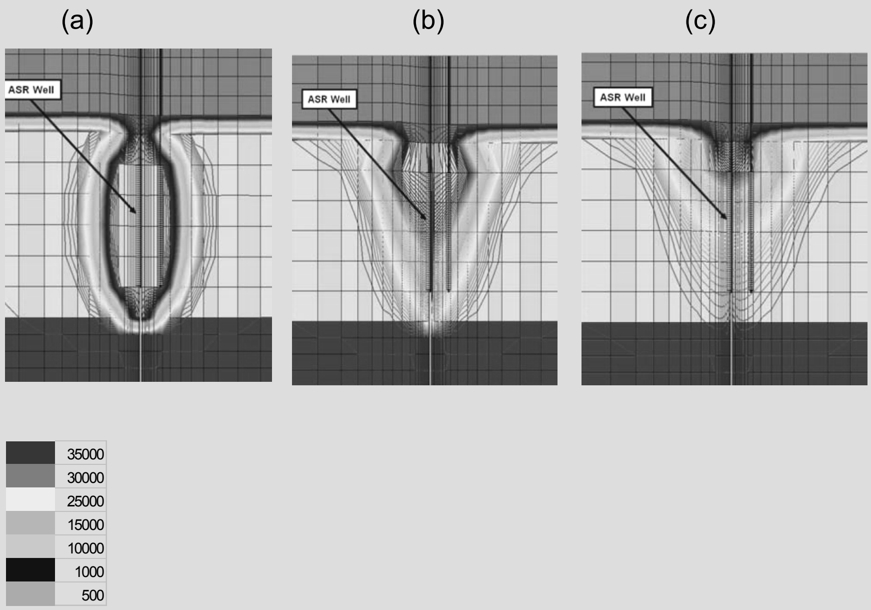
A



B

(A) Horizontal mesh resolution and (B) conceptual geology and vertical mesh resolution of numerical model for an ASR well.

BOX 3-5 Continued



(C) Cross sectional views of the concentration profile in the vicinity of the ASR well: (a) time = 30 days, at the end of the recharge phase; (b) time = 335 days, at the end of the storage period; and (c) time = 365 days, also at the end of the storage period.
SOURCE: Brown et al. (2006).

and flows; (2) determining regional changes in aquifer water quality TDS, sulfate, and chloride; (3) estimating groundwater recharge, discharge, and storage at larger spatial scales; (4) assessing the cumulative effects of existing and proposed water resource uses and developments; (5) assessing regional impacts on existing well users of the aquifer; and (6) evaluating the cumulative effects of various water management scenarios on water resources (Mack, 2003). Development of a regional model for the MUS system can be used to assess potential impacts related to a full-scale implementation of MUS, specifically ASR, and the feasibility of its development on such a scale. In the case of ASR, these models are valuable tools for (1) increased potential for saltwater intrusion caused by ASR pumping; (2) ASR well cluster site selection; (3) ASR well cluster design, layout performance including estimating recovery efficiency; (4) ASR well site evaluation of pressure- induced changes; and (5) localized ASR well pump design (dependent upon the appropriate model resolution).

When conducting a regional study, it is advised that model development begin early in the process. Model development can begin as soon as the preliminary hydrogeologic framework has been assembled. In this way, the model can help define a framework to study the system dynamics and organize field data as they become available. This type of analysis can occur before the calibration and sensitivity have been completed (Anderson and Woessner, 1992). Although the objective of the final model is to simulate density-dependent flow, the initial regional-scale modeling effort employing available codes—even those that are only capable of simulating constant density flow—can be useful. This effort may assist in guiding the test-drilling program (e.g., APTs) and associated data acquisition (e.g., monitoring). The described adaptations to the model development process can be thought of as an iterative process in which the modeler is continually updating and improving the model throughout the duration of the regional study. As additional data are integrated in space and time, the regional modeling tool becomes more valued as a tool that has potential to extrapolate findings over large areas and long periods of time. Furthermore, the regional model can begin to identify areas where local scale modeling may be preferred or required.

Local models are required when the spatial resolution of a regional model is not sufficient to capture changes in the storage zone. Unlike regional models, local models provide an in-depth analysis of local hydrogeologic properties for specific areas of interest within the regional model. Generally speaking, a regional model may have a resolution of 250 to 5,000 feet, while a local model may have a resolution of 50 to 250 feet. In local models, more emphasis would be placed on small groups of ASR wells, which would allow for a better understanding of the performance and impact of these wells within the MUS systems. The feasibility of injecting, storing, and recovering specified volumes of water at individual ASR wells and in local clusters of ASR wells is the issue to which the local models are most specifically directed. A horizontal variable resolution of a regional model should be developed at a resolution of 100 to 250 feet around the proposed ASR wells, increasing to 10,000 feet in the far reaches of the study area. Use of finite-element code would benefit from its geometry advantages such that the model mesh would have varying resolution across the model domain.

Overall, it is common practice to use both regional and local modeling tools because of the need to assess the system-wide impacts as well as other impacts from the local, complex geometry (e.g., dual porosity, changes in the storage zone). Regional models can assess potential impacts when the MUS system is being implemented. Local models can assess potential impacts of fracturing and changes in the storage zone. Monitoring with modeling can complement and strengthen the regional study as a whole. Depending on the project, modeling tools can be implemented to help define the storage zone, buffer zone, and native groundwater area in conjunction with technical experience, especially since deep monitoring is expensive. As in any modeling effort, it is advisable to start collecting data early, start model development early (including running sensitivity analyses to determine data gaps), and practice the iterative process of adapting the modeling tool to newly acquired data.

IMPACTS AND CONSTRAINTS OF THE MUS SYSTEM

Local, Intermediate, and Regional Flow

Local, intermediate, and regional flows can affect groundwater movement through an aquifer. Factors affecting the amount of local versus regional flow include topography, climate, nature and extent of unconfined and confined aquifers, hydraulic conductivity of the aquifer and confining layers (Johnston, 1997), and recharge patterns. Local flow systems are dominated by a topographically high recharge area and a topographically low discharge area (Schwartz and Zhang, 2003). Groundwater flow through unconsolidated sandy aquifers is usually the result of local flow systems (*http://capp.water.usgs.gov*). Johnston (1997) argues that under natural conditions, the amount of local flow can be much greater than regional flow. In addition, the flow velocity in local flow systems is usually greater than that in regional systems due to nearby streams (Schwartz and Zhang, 2003). As such, local flow systems are highly dependent on topography and recharge patterns.

A system with various topographically low sections between recharge and discharge zones is recognized as having intermediate flow (Schwartz and Zhang, 2003). Basin-filled aquifers, such as the California Central Valley, have intermediate flow that is representative of regional flow systems (*http://capp.water.usgs.gov*). In contrast, regional flows are dominant in topographically extensive low areas under natural conditions. In a regional flow system, changes in hydraulic conductivity can impact the vertical and horizontal hydraulic head of the aquifer. For example, in an aquifer with increasing hydraulic conductivity in a lower layer and a consistently lower hydraulic conductivity in a higher layer, the vertical hydraulic gradient would be greater than the horizontal hydraulic gradient in much of the system (Schwartz and Zhang, 2003).

Karst aquifers can be described by high heterogeneity resulting from groundwater flow, large voids, and high flow velocities (Bakalowicz, 2005). Groundwater flow through karst aquifers can be affected by extensive fracture systems, which increase total porosity and permeability (Herrera, 2002). In karstic environments, Bakalowicz (2005) makes a distinction between local and regional flows, indicating that conduit patterns depend on porosity and recharge and direction of the hydraulic gradient and the drainage planes, respectively. It must be noted, however, that secondary porosity can affect regional flows by increasing the hydraulic conductivity (Herrera, 2002). Karstic aquifers generally favor rapid flow and transport of solutes.

With a recharge basin, vertical flow beneath the basin becomes dominant. A groundwater mound will be developed during the recharge, but it can also reverse local flow direction against regional flow direction. Vertical flow can potentially increase baseflow in neighboring streams. With a vadose zone well, a groundwater mound will be developed beneath the well. However, its effect on the regional flow is minimal due to limited recharge capacity. Vertical flow is a primary component beneath the well, while horizontal flow becomes dominant after water reaches the water table. With well recharge, a groundwater mound will be developed surrounding the well. It can reverse the local flow direction relative to the regional flow, depending on the recharge capacity (Sheng, 2005). In general, horizontal flow is the dominating component, especially within a confined aquifer, even though a mound may be developed around the well.

For stressed aquifers in an arid or semiarid region, the MUS system can also restore local water levels created by historic groundwater mining or overwithdrawal within a wellfield and, in turn may, restore natural flow pattern of the regional aquifer. MUS in shallow aquifers can thereby have major effects on groundwater-surface water interaction. If surface water is well connected hydraulically to the shallow aquifer, a rise in the water table will likely increase groundwater flow into local streams, lakes, and wetlands via seeps and springs. In the case of wetlands, this could potentially have a major effect on water budgets (and, therefore, water depth) and nutrient budgets. The combination of changing depth and water chemistry might alter the aquatic ecosystems substantially. The loss of groundwater to the surface environment might also raise legal questions as to the ownership of the discharged water (Chapter 5). Conversely, if surface water upstream is diverted from streams and lakes for recharge, it may not only affect downstream flow, but also cause water quality deterioration. Such complications underscore the importance of both having a clear understanding of the hydrogeologic system and keeping MUS in the context of other water management activities and tools (Chapter 7).

Water Density: Uniform Density Versus Variable Density

Groundwater flow in an aquifer can be caused by density differences (Cserepes & Lenkey, 2004). Water with dissolved solids such as seawater is denser than fresh water, and as such, density calculations are imperative in estimating flow directions (Boulding and Ginn, 2003). Water's fluid properties vary with temperature, and hydraulic conductivity is affected as a result (Driscoll, 1995). To compensate for density effects on flow in modeling, flow equations can be adjusted to account for a variable-density fluid by including measurements of fluid pressure, intrinsic permeability, dynamic viscosity, and elevation (Ingebritsen & Sanford, 1998).

In an aquifer, dispersion and diffusion of solutes are critical to the effectiveness of MUS, especially for ASR. Buoyancy stratification allows injected water in a high-permeability zone to swell under the overlying confining unit floating atop the native, more saline groundwater due to its higher density (Merritt, 1985; Vacher et al., 2006). Multiple "wedges" can result from injection into low-permeability beds within heterogeneous storage zones (Vacher et al., 2006). Maliva et al. (2006), recognizing recovery issues associated with density-driven fluids, suggest a dual-zone approach to ASR whereby the open interval includes the storage zone; however, a second well in the upper part of the storage zone is used for recovery, taking advantage of the stratification effect. Alternatively, they suggest use of a "flapper valve" that would allow recharge through the entire open interval, but only recover within the upper part of the storage zone.

The effects of diffusion are highly evident in zones of lower permeabilities (Kasteel et al., 2000). Aquifers with high clay contents and native water salinity can greatly decrease their recovery efficiency as illustrated by Konikow (2001). Recovery efficiency is dependent on formation properties. Salinity, permeability, and thickness of the aquifer can affect recovery efficiency. Konikow (2001) indicated that the potential for clay dispersion is greatest in aquifers with swelling-type (2:1 clay lattice) clays. A successful ASR in coastal systems with brackish water is possible if flow patterns during recharge and withdrawal are constant (Konikow, 2001). Changes in the pattern can negatively impact an ASR. Solute concentration can create buoyancy forces greater than the fluid velocities created by hydraulic forces (Ingebritsen & Sanford, 1998).

Temperature

Increases in fluid temperature can result in a decrease in density, which in turn causes molecules to move faster (Fitts, 2002). Driscoll (1995) defines viscosity as the degree of resistance of a liquid to an applied force. A temperature increase causes a fluid's viscosity to decrease (Fitts, 2002). As a result, hydraulic conductivity increases by 50 percent between 10 and 26°C (Zijl and Nawalany, 1993). As groundwater flows through an aquifer, its temperature changes despite its high specific heat capacity (Boulding & Ginn, 2003). The natural geothermal gradient leads to an increase in temperature with depth of about 1°C per 20-40 meters (Bear, 1979).

If a significant temperature difference exists between native groundwater and recharge water, the MUS system will affect the hydraulic gradient between the native and recharge water. The size and shape of the storage zone could then be affected depending on the direction and magnitude of the temperature gradient. In addition, the MUS system could also cause changes in temperature for spring flow, which may be a critical parameter for sustaining a neighboring ecological system.

Aquifer Matrix

Physical Impacts

Land subsidence is a gradual settling or sudden sinking of the earth's surface owing to subsurface movement of earth materials (Galloway et al., 1999). More than 80 percent of the identified subsidence in the United States is a consequence of our exploitation of underground water, and the increasing development of land and water resources threatens to exacerbate existing land subsidence problems and initiate new ones. Subsidence is virtually an irreversible process, and cracks and fissures may coexist at the land surface as results of aquifer movement (Helm, 1994; Holzer, 1984; Sheng, et al., 2003).

Decrease in artesian head in compressible confined aquifer systems results in increased effective stress (grain-to-grain load) on the confined sediments. The magnitude of subsidence depends on the magnitude of change in head and on the compaction characteristics and thickness of the sediments. The greater the number of clayey beds in the aquifer system, the greater may be the compaction. Continuous measurement of compaction of materials in deep holes indicates rapid response to head change at most places in the subsiding areas. Subsidence can be slowed down or stopped by raising the level of the potentiometric surface or the artesian head sufficiently. One of the methods for controlling land subsidence is artificial recharge, which injects water into the stressed aquifer to raise the hydraulic head and curtail ongoing subsidence resulting from overwithdrawal of groundwater (ASCE, 2001). Observed subsidence and uplift after recharge in Santa Clara, California, demonstrates deformation of aquifer materials as results of groundwater pumping and artificial recharge (Schmidt and Burgmann, 2003). During cycles of recharge-storage-recovery of an MUS system, resulting subsidence and uplift may impact system operations and cause possible damages of infrastructure in the vicinity of the system. Li (2000) and Li and Sheng (2002) developed conceptual models to assess impacts of different scenarios of cycle loading of the ASR system on aquifer materials and concluded that confining units, especially clay layers or interlenses, will deform and result in additional subsidence during recovery of the stored water, and partial recovery of subsidence or seasonal uplift is also expected under a favorable condition even with an ASR system to retain groundwater levels.

The magnitude of subsidence associated with water-level decline appears to be related in large part to geologic factors such as (1) differences in mineral composition, (2) particle size, (3) sorting, (4) degree of consolidation, (5) degree of cementation, and (6) degree of confinement of the deposits in the groundwater reservoir. Thus, the ratio of subsidence to head decline will vary between groundwater reservoirs and even within a single groundwater reservoir. For example, measured ratios of subsidence to head decline vary from 0.008 to 0.1. Annual measured rates of land-surface subsidence in groundwater reservoirs range from a fractional value to about 0.5 m (1.5 feet) (ASCE, 1987).

Allowed Change in Hydraulic Head

Development and operation of a groundwater basin or aquifer in an area subject to alternating periods of drought and surplus suggests utilization of the groundwater storage during periods of deficient supply and the subsequent replenishment of the storage during periods of surplus (ASCE, 2001). In this context, these operations will occur in much the same manner as a surface reservoir would be operated. This will result in an artificial lowering of the water table or potentiometric surface during periods of deficient supply and a consequent return to former conditions during periods of surplus.

The actual limit or allowable range of fluctuation of the water table aquifer is primarily a matter of economics and aquifer characteristics. During wet periods, levels should not be permitted to rise so high as to cause waterlogging or property damage. On the other hand, levels cannot be drawn down to the point where it is economically impossible to extract and utilize the supply. Moreover, excessive drawdown can deteriorate the storage capacity of the aquifer by compaction, reduction of pore connections, or in extreme cases even the collapsing of structures. When excessive drawdown occurs, the costs of deepening existing wells, resetting pumps, or drilling new wells, in addition to the cost of obtaining new pumping equipment, are particularly important. In addition, environmental factors such as reduced baseflow to surface water bodies, spring discharge, and increased potential for sinkhole formation could also result from excessive drawdown. Excessive replenishment of the aquifer during surplus periods can also be undesirable because excessive recharge may result in hydraulic fracturing. Hydraulic fracturing is a result of increased fluid pressure over stress and rock strength (Domenico and Schwartz, 1998; Ingebritsen and Sanford, 1998).

Brown et al. (2005) identified four failure mechanisms related to hydraulic fracturing. These mechanisms include regional-scale shear failure of the rock matrix, hydraulic failure of the rock matrix, pore volume increase due to microfracture formation, and localized stress concentrations around the wellhead. The failure mechanisms—whether on a regional scale or localized—may limit or prevent implementation of an MUS project, since any one of the mechanisms could lead to the formation of preferential flow paths in the confining unit (Brown et al., 2005).

In April 1999, the Comprehensive Everglades Restoration Plan (proposed large-scale development of ASR facilities as the preferred method of providing additional freshwater storage required for overall restoration success (USACE/SFWMD, 1999). The proposed CERP system includes a total of 333 ASR wells and related surface facilities at the general locations. All proposed ASR wells have a target capacity of 18,927 m^3/d (5 Mgal/d) with water treatment facilities included in the conceptual CERP ASR components. The total cost of the proposed CERP ASR system is approximately $1.7 billion, about one-fifth of the total estimated cost of the CERP.

In cooperation with a multiagency project delivery team, the U.S. Army Corps of Engineers and the South Florida Water Management District were tasked with evaluating the feasibility of the proposed CERP ASR projects individually and through the development of a regional feasibility study. A component of the *ASR Regional Study*, outlined in Brown et al. (2005), was to determine the pressure induced effects of an anticipated daily ASR recharge volume of 1.67 billion gallons, to the upper Floridan Aquifer System (FAS) and overlying Hawthorn Group sediments, specifically, the effects on piezometric pressure and hydraulic fracturing potential.

The magnitude of the increase or decrease in piezometric pressure within the upper FAS during recharge and recovery cycles is highly dependent on numerous factors such as aquifer transmissivity, well spacing, and aquifer porosity. During ASR recharge, increases in static piezometric (hydraulic) head of 30.48 to 60.96 m (100 to 200 feet) near the pumping wells are certainly possible based on both analytical and numerical models. Conversely, during ASR recovery, decreases in static head of similar magnitudes are possible. These anticipated pressure changes may present planning and engineering constraints that somewhat limit ASR development.

The most important of these constraints is the potential for hydraulic fracturing of the limestone rock of the upper FAS or overlying Hawthorn Group sediments. Brown et al. (2005) concluded that microfracturing of the FAS limestone due to dilatancy may occur at a total hydraulic head greater than 183 feet. In addition, during recovery operations, settlement or subsidence of the overlying Hawthorn Group clays is a possibility that requires further examination. Conversely, during ASR recharge cycles, expansion or "lengthening" of the Hawthorn Group clay is a remote possibility.

For ASR design purposes, the pressure changes will also likely constrain wellhead design or pump selection. In addition, continually injecting at high pressures could result in high costs for electricity. Brown et al. (2005) noted that the estimated electricity cost for the 333 planned ASR wells would surpass 50 percent of the entire operation and maintenance budget for the CERP ASR program. FAS heads substantially higher than the current regional flow system could also lead to changes in flow direction or velocity.

Slow regional subsidence poses serious problems in the operations of many types of engineering structures, particularly those involved in the storage, transport, and pumping of water. For example, tilting of the land surface can appreciably reduce the flow of water in low-gradient gravity canals. This has occurred in the United States Bureau of Reclamation's Delta-Mendota Canal along the west side of the San Joaquin Valley in California. In addition, smaller structures such as drains and sewers can be affected, and even the channel capacity of streams may be altered. Tilting can also affect the operation of pumping plants because such plants may be highly sensitive to minute tilting of the land surface. In a critical area such as a coastal bay where bordering lands subside, levees may have to be built. This has been done in the southern San Francisco Bay area in California to prevent flooding of adjacent agricultural, urban, and industrial areas by saline bay waters (Fowler, 1981). In addition, when the consolidating sediments are deep, casings of the water wells are compressed and frequently ruptured, requiring expensive maintenance and replacement. As sediments continue to compact in a groundwater reservoir, reduction in groundwater storage capacity and even in the permeability of the sediments may occur, although the usual case is for the fine grained materials (primarily clays) to consolidate rather than the more important granular materials of the aquifers. The legal aspects of land surface subsidence caused by groundwater withdrawal are additional concerns facing water resource managers (Kopper and Finlayson, 1981).

Chemical Impacts

Hydrogeochemical and biogeochemical reactions may affect physical aspects of the aquifer and MUS system performance. Clogging via microbial activity or mineral precipitation, for example, reduces hydraulic conductivity and affects MUS system performance. Mixing of water during MUS activities may lead to dissolution and enhancement of dual porosity. Although these processes are described in this chapter, significantly more detail is provided in Chapters 4 and 6.

CONCLUSIONS AND RECOMMENDATIONS

Conclusion: To facilitate the siting and implementation of MUS systems, maps of favorable aquifers and hydrogeological characteristics can be prepared using 3D capable geographical information systems (GIS). At a regional or statewide scale, such GIS maps can help visualize and characterize major aquifers for future development of MUS systems, map and analyze regional changes in head and flow patterns, and facilitate comprehensive, regional water resources management. At a project scale, they can aid in establishing the design, spacing, orientation, and capacity of wells and recharge basins, evaluating their impact on the environment and existing users, estimating the critical pressure for rock fracturing, visualizing the movement of stored water throughout the system (especially useful for systems with waters of varying density or quality), and evaluating the extent of potential water quality changes in the aquifer during storage and movement.

Recommendation: States, counties, and water authorities considering MUS should consider incorporating 3D capable GIS along with existing hydrogeologic, geochemical, cadastral, and other data in (1) regional mapping efforts to identify areas that are, or are not, likely to be favorable for development of various kinds of MUS systems, and (2) project conception, design, pilot testing, and adaptive management.

Conclusion: Long-term local and regional impacts of MUS systems on both native groundwater and surface water have been recognized, including changes in groundwater recharge, flow, and discharge, and effects on aquifer matrix such as compaction of confining layers or clay interlayers during recharge and recovery cycles.

Recommendation: Monitoring and modeling should be performed to predict likely effects—positive or negative—of MUS systems on the physical system, including inflows, storage, and outflows. Appropriate measures can and should be taken to minimize negative effects during operations.

Conclusion: Groundwater numerical modeling at regional and/or high-resolution local scales provides a cost-effective tool for planning, design and operation of an MUS system.

Recommendation: Analyses using groundwater flow and solute transport modeling should become a routine part of planning for, designing, and adaptively operating MUS systems. Uncertainty analysis should also be incorporated into prediction of a system's short- and long-term performance, especially regarding the expected values of recovery efficiency and storage capacity.

Conclusion and Recommendation: In addition to the topics above, research is particularly needed, and should be conducted, in the following areas:

- *Hydrologic feasibility.* This includes (1) lack of knowledge about storage zones and areas favorable for recharge for major aquifers in the United States; (2) limited understanding of how aquifer heterogeneity, scale effects, and other physical, chemical, and biological properties impact recharge rate and recovery efficiency of the MUS system; (3) lack of understanding of matrix behavior, especially fractured aquifers, during recharge versus withdrawal tests (e.g., expansion vs. compaction) to prevent or limit artificially induced deformation of the aquifer matrix; (4) need to develop of tools to analyze non-Darcian flow around recharge wells to avoid poor design of recharge wells; and (5) need for overall characterization, system recovery efficiency, optimum placement of monitoring wells, recharge and pumping impacts, and hydraulic fracturing in an aquifer with dual porosity.
- *Impacts of MUS systems on surface water*. How, in terms of both quantity and timing, might a surface spreading or well recharge facility affect the flow of neighboring streams? What would be the hydrologic, ecological, and legal consequences of this interaction between the MUS system and surface water? An integrated or system approach should be developed and employed for assessing such impacts.
- *Technology enhancement and methodology development for determining hydrological properties of the aquifers and their impacts on performance of the MUS system.* These include (1) surface and borehole geophysical methods to determine hydrological properties and the extent of recharge water volumes during cycle testing; (2) optimization of cycle test design (frequency, duration, and intensity) to improve performance of MUS systems for various hydrological settings; (3) better conceptual models for delineation of storage zone and recovery zone; and (4) better understanding of non-Darcian flow near recharge wells through experimental study and field monitoring, and further development of theories and numerical models to assess the interaction of stored water (especially urban runoff) with native groundwater.

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4
Water Quality Considerations

INTRODUCTION

Water quality is characterized by the chemical (organic and inorganic), physical, and microbiological nature of the water. The monitoring and testing that go along with this characterization must focus on both constituents of concern to human health and those that affect operations of the water systems. The development of a system for managed underground storage (MUS) of water involves the testing and characterization of the source water, the aquifer geochemistry and native water quality, the stored water, and the recovered water. The subsurface has the capacity to attenuate many chemical constituents and pathogens via physical, chemical, and biological processes. Critical to MUS is an understanding of the mixing of often chemically and microbiologically different waters, which may react with each other and with materials comprising the aquifer matrix. The reactions that occur can ultimately improve or diminish the stored water quality chemically and microbiologically. Water quality changes can be variable in both space and time. Furthermore, among the potential suite of reactions are those that can cause clogging or dissolution of the aquifer matrix and so affect MUS operation. The consequences of the potential reactions during storage underscore the importance of a comprehensive aquifer characterization to fully understand the water quality changes that may occur during MUS. An understanding of temporal changes in the quality of water prior to and during storage is critical and is intertwined with the application, treatment requirements, and use of the water after it is recovered. This understanding may also influence the treatment of waters prior to storage. "Successful" MUS is therefore much more than a function of effective hydrologic engineering; MUS must also consider the broad spectrum of processes—microbiologic, hydrochemical, geochemical, and hydrogeologic—as they influence water quality and performance of the system.

The mix of constituents in source waters for MUS varies, depending on the natural purity of the water and constituent inputs and modifications through human activities (e.g., agricultural, industrial, commercial, and residential land use, engineered treatment processes). Public concerns about these constituents may vary depending on whether the classification is "health-related" or "aesthetic." The purposes of this chapter are to describe: (1) the range of constituents in MUS waters; (2) hydrogeochemical and microbiological processes involved as source waters interact with the native ground water and rocks or sediments comprising the aquifer, and the impact of these processes on MUS performance; and (3) predictive tools for water quality and aquifer changes.

CONSTITUENTS IN WATERS THAT CAN AFFECT PERFORMANCE AND OPERATION OF MUS

Constituents

Two overlapping sets of water quality parameters are important to MUS performance and so must be considered in designing MUS systems. Constituents regulated in drinking water (as described by the Safe Drinking Water Act [SDWA]) comprise a well-defined list with concentrations that must be met in drinking water supplies for either human health or aesthetic reasons. While the SDWA prescribes the list of both chemicals and microorganisms that have been the primary impetus for water quality goals, this list is not sufficient to evaluate the quality of the various waters (source water, native groundwater, stored water, etc.) for an MUS system. In order to establish a sustainable MUS system, constituents that lead to aquifer clogging or dissolution, or other reactions that improve or degrade water quality during MUS operations must also be evaluated. The constituent concentrations that are important for operations are not embodied in a regulatory list, but emerge from consideration of the reactions that can impact MUS performance and the particular type of MUS system (e.g., type of source water, recharge method, native groundwater characteristics, and aquifer geochemistry). Importantly, the microbial and chemical water quality can improve or degrade during any stage of MUS.

The list of contaminants developed under the SDWA includes the list of chemical and microbiological constituents that have established legal enforceable maximum contaminant levels (MCLs) and/or treatment technology requirements and MCLGs (maximum contaminant level goals). Total coliform bacteria are used from a regulatory monitoring perspective to judge drinking water microbiological safety. There is also emerging concern about "new" (previously unmonitored) chemicals and constituents that occur in water as a consequence of human activities and are not regulated (e.g., endocrine disrupting chemicals, pharmaceuticals, personal care products). For many of the chemicals in this classification, analytical techniques appropriate for environmental samples are relatively new and complex. The World Health Organization also has developed a list of constituents of interest in water for health goals that includes some compounds that are not regulated by the U.S. Environmental Protection Agency (EPA) including, for example, the cyanobacterial toxins that can be found in surface waters.

To fully appreciate the broad water quality characteristics found in MUS systems from the ambient groundwater to the source, stored, and recovered water, the physical, chemical, and microbiological water quality constituents need to be understood and measured. These are described briefly in the following sections, and extended descriptions are available in Appendix A.

Physical Characteristics

The first impressions of water quality are often based on visual observations. Water is expected to be free of particles (turbidity), color, taste, and odor. ***Turbidity*** may increase clogging, and these particles can also harbor pathogens and enhance their survival in the presence of a disinfectant. ***Color*** is often the result of dissolved organic matter, for example, humic and fulvic acids. ***Taste*** is often related to the presence of iron or manganese in the water. It may also be due to high levels of chlorine used as a disinfectant. ***Odor*** may be caused by decomposition of organic matter or reduction of dissolved sulfate; the control of odors is among the priority issues with respect to public acceptance of a project.

Additional important physicochemical characteristics of MUS waters include dissolved oxygen, pH, oxidation-reduction potential (Eh), specific conductance, and temperature. ***Dissolved oxygen*** (DO) is required by any aquatic organisms that respire aerobically (i.e., breathe oxygen). The presence of DO tends to minimize odors, but it may cause oxidation of sulfide minerals or organic matter in aquifers that can lead to the release of arsenic and other metals. The DO content of recharged water is affected by temperature and so can vary significantly with the season. Dissolved oxygen saturation (with respect to atmospheric oxygen content) is a strong function of temperature within the relevant environmental range. For fresh water (< 2000 mg/L of total dissolved solids [TDS]), the oxygen saturation ranges from approximately 7 mg/L at 35^{o}C to 12.8 mg/L at 5^{o}C. Water treatment processes, such as ozonation and chlorination, also affect the DO. The ***pH*** is a measure of the hydrogen-ion concentration, or the acidity, of water. It influences everything from the ability of a mineral to adsorb toxic metals to the dissolution of the aquifer materials. ***Oxidation-reduction potential (ORP or Eh)*** is another critical parameter because it indicates processes such as iron dissolution or precipitation and proportions of various dissolved nitrogen species such as ammonia. Along with pH, Eh provides a measure useful for gauging conditions that favor the persistence of certain organic contaminants or the survival of certain pathogens. ***Specific conductance*** is a measure of how well a given water sample conducts an electrical current and can give a good estimate of the TDS in a solution. Finally, ***temperature*** affects the speed (kinetics) of chemical reactions in the subsurface, whether they are mediated by bacteria or not.

Organic Constituents

Four classes of organic constituents are particularly important to MUS systems: total organic carbon, disinfection by-products, other regulated organics (aside from disinfection by-products), and so-called emerging contaminants. ***Total organic carbon*** includes both dissolved organic carbon (DOC) and particulate organic carbon (POC) and is composed primarily of natural organic matter (NOM). DOC can lead to the formation of disinfection by-products. In addition, the degradation of labile dissolved and particulate organic carbon in recharge water can lead to reductions in DO, ORP, and pH and can also cause clogging through stimulation of biomass growth. ***Disinfection by-products***, or DBPs, are formed as a consequence of reactions between disinfection chemicals (chlorine, chloramine, and ozone) used to treat microbial pathogen contaminants and DOC. They are often small, halogenated (e.g. chlorinated, brominated) or nitrogen-containing organic compounds. Because the precursor organic matter is of variable composition, the DBPs produced encompass a spectrum of chemicals including the regulated trihalomethanes (THMs) and haloacetic acids (HAAs). Regulated trace organic contaminants, such as petroleum hydrocarbons, chlorinated solvents, and regulated pesticides, are known toxins or carcinogens and are problematic in thousands of contaminated sites around the country. Their behavior must be considered for any particular MUS if they are present in either the source water or the groundwater system. Unlike DBPs, these chemicals are not created in situ. Methods to monitor these chemicals in drinking water supplies are well established and routinely available. The fate and transport of these chemicals in groundwater are relatively well understood (compared to emerging contaminants) as a consequence of prior groundwater studies. The behavior of these compounds in standard water treatment facilities is also well known. For these reasons, the discussion of this group of contaminants in this report is limited, and the reader is referred to more comprehensive reviews. ***Emerging contaminants*** are any synthetic or naturally occurring chemicals or microorganisms that are not commonly monitored in the environment but have the potential to enter the environment and cause known or suspected adverse ecological and/or human health effects (*http://toxics.usgs.gov/regional/emc/).* They are widespread and include antibiotics and other pharmaceuticals, personal care products, hormones, and many other compounds.

Inorganic Constituents

Inorganic chemical constituents of concern in MUS source waters can be grouped as nutrients, nonmetals, and metals and metalloids. Nitrogen and phosphorous species are known as ***nutrients*** because they are essential for the growth of microorganisms and plants. However, they can also contribute to deleterious growth of algae or microorganisms in MUS systems. Nitrogen is soluble in several forms, including nitrate and nitrite. Phosphorus is generally poorly soluble as phosphate. The ***nonmetals*** of concern include species such as chloride and sulfate and occasionally borate. Typically, these are part of a larger problem of salinization either in the case of recharge into brackish groundwater or due to evaporation in arid regions. The ***metals and metalloids*** of concern are often present at trace concentrations, and many are classified as priority pollutants. Examples of these include arsenic, cadmium, mercury, lead, and chromium. They are associated with a wide variety of problems from developmental delays in children to various cancers, bone disease, and skin problems. Radionuclides of greatest concern are uranium and radon, both of which are carcinogens. Iron and manganese, except at very high levels, are primarily of concern because they influence the aesthetic quality of the water. Iron can be related to clogging problems as well.

Microbial Constituents

Important human pathogens for MUS systems are those microorganisms including bacteria, parasites, and viruses that come from both human and animal fecal pollution and naturally-occurring microorganisms that reside and grow in the aquatic environment such as cyanobacteria (toxic algae) and *Legionella.* Often the distinction between human and animal sources using microbial source tracking techniques is advantageous with regard to developing strategies to control the source. In the United States, waterborne outbreaks (common-source epidemics associated with contamination of the drinking water) have occurred in both community and non-community systems. Groundwater was the supply most often associated with these outbreaks (compared to springs, surface water, or contamination of the distribution system) often because disinfection was inadequate or not used to treat microbially contaminated wells (Liang et al., 2006). From 1989 to 2002, 64 percent of drinking water outbreaks were from a groundwater supply, and more recently from the 2001 to 2002 and 2003 to 2004 reports, groundwater was associated with 92 percent and 52 percent of the drinking water outbreaks, respectively (Blackburn et al., 2004;Liang et al., 2006). Bacteria, including fecal bacteria such as *Campylobacter* (associated with animal and human wastes) and aquatic (nonfecal) bacteria such as *Legionella* as well as enteric viruses from human fecal wastes, were the most common causes of the illnesses.

Native Groundwater and Aquifer Geochemistry

Native Groundwater Geochemistry and Associated Aquifer Classification

Native groundwater quality in an aquifer is important to consider in planning an MUS system because it provides information about constituents likely to dissolve into stored water as it equilibrates with the aquifer matrix. Knowledge of native groundwater quality is also critical to evaluating the potential for chemical reactions occur as recharged and native waters mix in the transition zone. In addition, native groundwater chemistry provides a useful means for aquifer classification that is related to the aquifer mineral matrix.

In uncontaminated groundwaters, major ions typically originate from the weathering of aquifer minerals. Hence, there is a strong association between the major ions identified and the mineral composition of the aquifer. Major cations include Ca^{2+}, K^{+}, Na^{+}, and Mg^{2+}, and major anions include Cl^{-}, HCO_3^{-}, SO_4^{2-}, and sometimes NO_3^- (Table 4-1) (Freeze and Cherry, 1979; Hem, 1985). Concentrations of nitrate sufficiently high to warrant its inclusion as a major anion are generally attributable to anthropogenic influence. The fingerprint of the major cations and anions in groundwaters (e.g., their concentrations and relative proportions) can be used to distinguish among hydrochemical units in the subsurface. For example, aquifers comprised of limestone (mostly calcium and/or calcium-magnesium carbonate minerals) will typically exhibit calcium as the dominant cation and bicarbonate as the dominant anion. Table 4-1 summarizes some hydrochemical attributes typical of groundwaters contained within different types of aquifer rocks. This table generalizes compositions typical of potable aquifers that have low (less than 1,000-2,000 mg/L) TDS.

Although trace metals and metalloids in groundwater are often associated with contamination, they can also occur naturally in groundwaters as a consequence of water-rock interactions. Recent work (Lee and Helsel, 2005) suggests that background (without anthropogenic contamination) trace element concentrations of barium, chromium, copper, lead, nickel, molybdenum, and selenium have a 1.0 to 1.5 percent likelihood of exceeding federal drinking water standards. The authors report that arsenic is an exception, with a 7 percent likelihood of exceeding the federal drinking water standard.

Unlike trace metals, regulated organic contaminants occur in groundwater solely because of human activities. Regulated industrial chemicals occur in groundwater as a consequence of point source discharges via leaks, spills, or historical disposal. In addition, regional contamination of groundwaters can occur from nonpoint or widely distributed sources related to land use. Examples of such chemicals include pesticides and nutrients (Scanlon et al., 2005).

TABLE 4-1 Typical Major Ion Chemistry in Groundwaters Associated with Potable Aquifers in Different Types of Rock

Matrix	**pH[a]**	**Major dissolved species[b]**
Carbonate	Circumneutral to basic	Ca^{2+}, Mg^{2+}, HCO_3^-
Unconsolidated and consolidated siliciclastic sediments		
Siliciclastic; alluvium, glacial	Circumneutral to acidic	Ca^{2+}, Na^+, HCO_3^-; SO_4^2 ; mixed cation
Fractured Bedrock (igneous, metamorphic, brittle sedimentary)	Basic	Mg^{2+}, Ca^{2+}, Na^+, HCO_3^-; SiO_2

[a] more acidic near recharge areas.
[b] ions and dissolved chemicals (see glossary for definitions). Na^+, Mg^{2+}, Cl^- are generally higher proximal to saline water bodies and within deeper "formation" waters; NO_3^- in high-recharge areas and unconfined aquifers.
SOURCE: Freeze and Cherry (1979); Hem (1985).

The microbiological quality associated with bacteria that naturally reside in the system is not well studied. Those involved in biochemical processes or bioremediation have been the primary focus of in situ studies. Many of the bacteria are anaerobic or facultative aerobes. There is a large emphasis in the literature on groundwaters impacted by microorganisms of surface water or wastewater origin.

Regulatory Classification of a Potable Aquifer

In addition to the water chemistry-based classification system for aquifers described above, there exist regulatory aquifer classifications that define an aquifer as ”potable” or ”non-potable” or describe its relative vulnerability to surface sources of contamination. Although aquifers within either classification can be considered for MUS, the regulatory designation may affect operational requirements, particularly source water quality, for the MUS system. Chapter 5 further describes regulation pertinent to MUS.

Most aquifers are protected by generic antidegradation policies such that no anthropogenic activity can lead to a measurable or perceived decline in water quality. This is due partly to the fact that groundwater is more difficult to clean up once contaminated. Protection of a potable aquifer is a key consideration for an MUS system and is addressed through water quality monitoring associated with drinking water applications.

Federal regulations classify (or designate) potable aquifers based on the following criteria: current use of the groundwater, water availability, and water quality as indicated by total dissolved solids. It is presumed that an aquifer classified as an underground source of drinking water (USDW) will meet the coliform bacteria regulatory requirement (<1/100 ml), yet the Ground Water Rule (*http://www.epa.gov/safewater/disinfection/gwr*) now recognizes the need for disinfection of groundwater used for potable purposes. Specific regulatory text describing an underground source of drinking water is provided in Box 4-1.

By law, state water quality regulations are at least as stringent as federal regulations. As a result, potable aquifer designations in some states are more detailed or involved than the federal regulation requires. Florida is among the many states that provide examples of additional regulatory classifications for aquifers. The Florida code defines three categories of aquifers for potable use based on the TDS of water in the aquifer and whether the aquifer serves as a single source of drinking water. It also lists two nonpotable use classifications for aquifers with high TDS for which there is no reasonable expectation that the aquifer will serve as a source of future drinking water. Confined aquifers so classified may be used for wastewater injection.

BOX 4-1
Federal Language Designating an Aquifer as 'Potable'

According to Section 144.3, Title 40, of the Code of Federal Regulations, an underground source of drinking water (USDW) "means an aquifer or its portion:

(a) (1) Which supplies any public water system; or
(2) Which contains a sufficient quantity of groundwater to supply a public water system; and
(i) Currently supplies drinking water for human consumption; or
(ii) Contains fewer than 10,000 mg/l total dissolved solids; and
(b) Which is not an exempted aquifer."

The same section states, "Exempted aquifer means an 'aquifer' or its portion that meets the criteria in the definition of 'underground source of drinking water' but which has been exempted according to the procedures in Sec. 144.7" (Title 40 of the Code of Federal Regulations).

Source Waters

Differences between the source water and native groundwater lead to reactions during storage that can impact recovered water and either improve or degrade its quality and/or impact MUS performance. To assess the potential for such reactions, evaluation of the source water quality is essential.

With a few important and notable exceptions, source water is the origin of most anthropogenic organic and microbial contaminants in stored groundwater. The exceptions include organic disinfection by-products that can be formed in the groundwater system through reaction of residual chemical disinfectants with natural organic matter. This statement also presumes that the groundwater system has not received contaminants through prior anthropogenic activities (e.g. spills, leaks, or nonpoint chemical use) that could contaminate the stored water.

Surface waters, other groundwaters (from interbasin or interaquifer transfers), urban stormwater runoff, and treated or reclaimed wastewater are all potential sources for MUS. Typical constituent classes of concern to MUS from a water quality perspective that are associated with different water sources are listed in Table 4-2. In many cases, it is mandated that the source water be treated prior to storage, with the treatment level often defaulting to creating water that meets drinking water standards. However, poorer-quality waters may be used. The feasibility of using lower-quality source waters depends on issues such as planned end use of the stored water, aquifer classification, post storage treatment, and in situ reactions that occur during recharge or storage. Use of such waters for recharge is also constrained by regulatory limitations. For those waters used for other purposes, the main concern may be potential or measurable water quality degradation in nearby groundwaters.

TABLE 4-2 Selected Constituents in Source Waters and Relative Concern for MUS[a]

Constituents	Untreated Ground-water[b]	Surface Waters	Urban Stormwater Runoff	Waters Treated to Drinking Water Standards	Wastewater Treated for Non-potable and Indirect Potable Use
Salinity	Low	Low or medium	Low to medium	Low	**High**
Nutrients (NO_3^-, etc.)	Medium	Medium	Medium	Low	**High**
Metalloids, including arsenic	Low to medium	Low	Medium to **high**	Low	Low
Mn, Mo, Fe, Ni, Co, V,	Low to medium	Low	Medium	Low	Low
Trace organics	Low to medium	Medium	**High**	Low	Medium
Total organic carbon (TOC)	Low to medium	Medium to **high**	Medium	Low	Medium
Disinfection by-products	Low	Medium	Low	**High**	**High**
Micro-organisms	Medium to **high**	**High**	Medium	Low	**High**

[a]The relative concerns shown in the table are based on committee consensus.
[b] Assuming source is a potable aquifer.

The case study in Box 4-2 illustrates a situation in Florida where stormwater is being used for groundwater resource augmentation. In addition, stormwater runoff has been used for groundwater recharge on Long Island, New York, and—mixed with other water types—in Orange County, California, for many decades. However, caution is always warranted with stormwater because of its highly variable chemical and microbiological nature. Even in the same location, the quality of stormwater runoff may vary with rainfall quantity and intensity, time since the last runoff event, and time of the year. Stormwater runoff from industrial areas, dry weather storm drainage flow, salt-laden snowmelt flow, construction site runoff, and flow originating from vehicle service areas are particularly problematical for artificial recharge (NRC, 1994).

There are promising new techniques to assess the risks posed by the use of stormwater. Page et al. (2006) used a Hazard Analysis and Critical Control Point (HACCP) framework to evaluate the viability of a potential ASTR project (see Chapter 6). They collected data on the number and types of industries in subcatchments, the likely chemicals used by these industries, stormwater quality, pollutants (and potential pollutants), operational procedures for stormwater management, barriers to hazards entering stormwater and control points for pollutant management. While their results generally supported moving forward,

BOX 4-2
Drainage Wells in Orlando, Florida

Since the early 1900s, drainage wells have been utilized for lake-level control and management of urban runoff. These wells are recognized as important components of groundwater resource augmentation and as such are now referred to as aquifer recharge wells. More than 400 of these wells divert approximately 30 million to 50 million gallons per day (Mgal/d) of lake overflow and stormwater runoff to the upper Floridan Aquifer System. The positive aspect of recharge wells is self-evident; however, concerns exist with regard to the introduction of untreated urban runoff (e.g., petroleum by-products, metals, nutrients, pesticides, and microbes) into the aquifer. Pre-recharge treatment strategies can be employed, including first-flush bypass, screens, filters, and disinfection systems.

The Central Florida Aquifer Recharge Project (CH2M Hill, 2006) was designed to assess these water quality concerns and potential strategies, specifically addressing the fate of bacteria in the Floridan Aquifer System, the effectiveness of passive stormwater treatment for reducing bacteria, and the effectiveness and cost feasibility of physically reducing bacteria in lake water recharge. These goals were addressed through (1) installation of monitor wells, (2) completion of groundwater tracer tests to confirm communication between the recharge and monitor wells, and (3) implementation of a comprehensive monitoring plan that includes broad-spectrum analyses of organic and inorganic constituents as well as microbes. During wet- and dry-season sampling, attenuation of nearly all constituents was observed. For example, up to a six-order-of-magnitude reduction in microbial concentrations was observed over a lateral distance through the aquifer of up to 450 feet. Arsenic, however, exhibited a statistically significant increase along the flow path between the recharge and monitor wells. A high degree of air entrainment during recharge, confirmed by borehole video, may have contributed to the release of arsenic from the aquifer matrix. The conclusions of this important and well-designed study were contrary to expected results. Metal mobilization was not anticipated, and initial concerns regarding microbes and synthetic organics were found to be uncorroborated. Based on the results of this study, government agency-sponsored random sampling of private wells is under way to assess elevated levels of arsenic.

they concluded that chemicals such as pesticides, herbicides, and endocrine disruptors, which were not monitored in real-time, required further research to validate that they were either absent or being removed effectively by the pretreatment system.

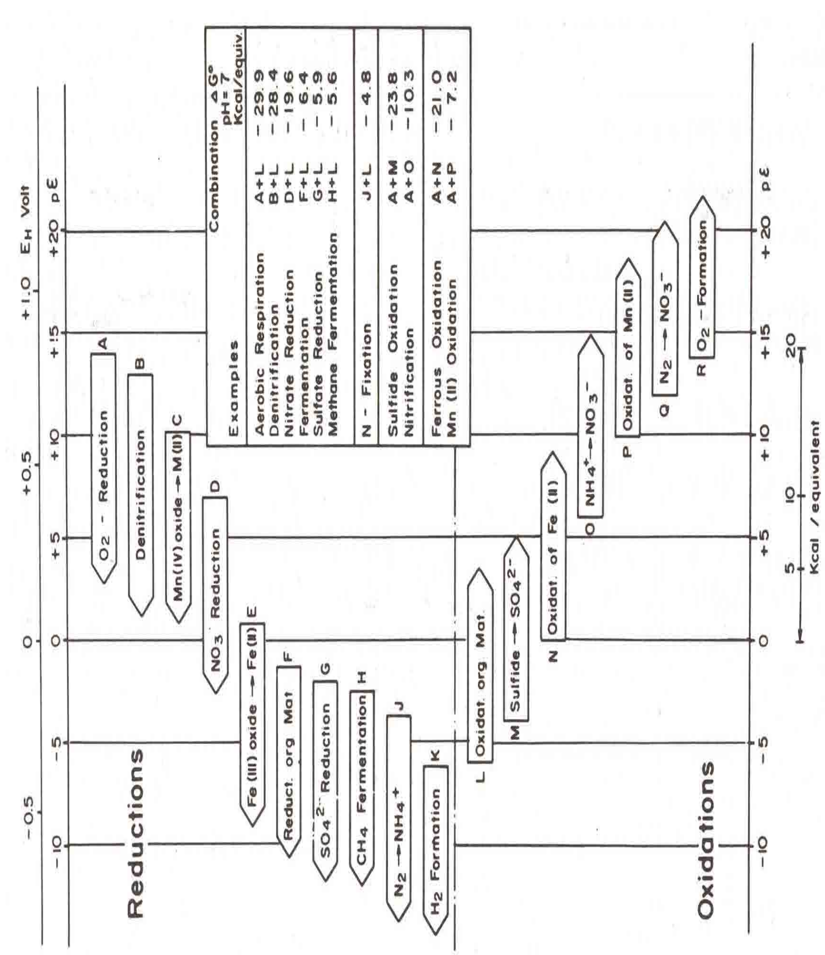
SUBSURFACE PROCESSES THAT AFFECT WATER QUALITY IN MUS SYSTEMS

Biogeochemical reactions, including water-rock interactions, that occur during MUS activities are dynamic in both space and time and are a consequence of mixing recharge water with water quality parameters that differ from the native groundwater in the aquifer. The reactions that occur result from mixing between native and recharged water, interaction between the recharged water and the aquifer media, and changing the environmental conditions of the recharged water (e.g., storing water underground that resided formerly at the surface and was open to the atmosphere). Departure from thermodynamic equilibrium among the recharged water, native groundwater, and aquifer media is the driving force for the changes in water chemistry and/or physical aquifer characteristics (e.g., permeability) that occur in the recharge zone. Chemical reactions that control or influence concentrations of contaminants during storage include oxidation-reduction (redox) reactions, acid-base reactions, sorption-desorption reactions including ion exchange, mixing (diffusion-dispersion or mechanical dispersion), and precipitation-dissolution reactions. Nearly all of the important reactions are mediated by common soil microorganisms native to the environment. Also, many of the most common (or important) geochemical processes that occur in situ encompass multiple reaction categories (e.g., redox, acid-base). Because of the high importance of redox reactions to water quality and aquifer integrity during underground storage, these are described in greater detail than the other reaction types. Detailed and rigorous discussions of each of these types of reactions in aqueous systems can be found in several texts, including (Drever, 1997; Langmuir, 1997; Stumm and Morgan, 1996)

Redox Reactions

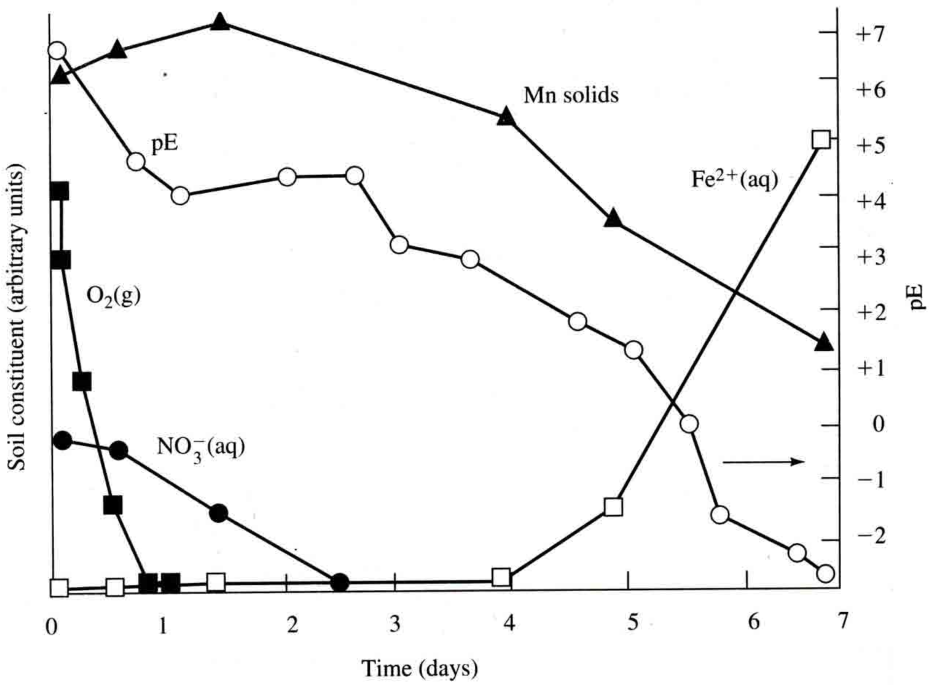
In a redox reaction, electrons are transferred between chemicals with a concomitant gain or release of energy. Species are termed oxidized if they are electron poor (e.g., nitrate, carbon dioxide, Fe(III) As(V)) and reduced if they are electron rich (e.g., nitrite, carbon in organic matter, Fe(II), As(III)). Only elements that can exist in multiple "electron" forms (species), such as carbon, nitrogen, arsenic, and iron, can participate in redox reactions. In a redox reaction, an oxidation reaction (in which one species loses an electron) must be coupled to a reduction reaction (in which one species gains an electron) because there exist no "free" (e.g., not part of an element) electrons. Although there are no free electrons within a system, the redox condition or potential of the system can be gauged by the dominant forms of redox- sensitive elements in the system and is often reported as the Eh or pε of the system. A lower value of Eh or pε indicates that the system is more reduced. Flowing rivers that are open to the atmosphere generally contain significant dissolved oxygen and are oxidizing. Many (but certainly not all) groundwaters have very low or immeasurable dissolved oxygen concentrations and have relatively high concentrations of more reduced species such as reduced iron (Fe^{2+}) or reduced sulfur (S^{2-}).

The redox reactions that occur during groundwater storage are typically exothermic (reactions that release energy). Microorganisms often mediate these reactions, which otherwise occur very slowly, and gain energy for growth. In general, microorganisms oxidize organic matter by utilizing available electron acceptor(s) to gain energy, and therefore, organic matter can serve as a driver of redox potential changes within a system. It can be either in the dissolved phase or as part of the aquifer solids. The energy available from coupling the oxidation of DOC to the reduction of different elements is quite variable (Figure 4-1A). In general, the most energetically favorable coupling available dominates a system.



A

FIGURE 4-1(A) Oxidation of organic matter coupled to reduction reactions. The greatest energy gain is associated with species toward the top of the figure. Changing availability of the terminal electron acceptors can result in dynamic spatial and temporal geochemistry. From Stumm and Morgan (1996) . Reprinted, with permission, Stumm and Morgan (1996). Copyright 1996 by John Wiley & Sons.



B

Figure 4-1(B) Relative changes in water chemistry and pε that occur as a consequence of the sequential use of electron acceptors in an inundated soil. SOURCE: Sposito (1989) as cited by Langmuir (1997). Reprinted, with permission, from Sposito (1989). Copyright 1996 by Oxford University Press.

Hence, the redox potential of a system depends on the type and quantity of available degradable organic matter and electron acceptor. For example, if the amount of degradable organic matter exceeds the available dissolved oxygen, which is a common occurrence in groundwater, then the system will become denitrifying *if* nitrate is available to be used as an electron acceptor. If no nitrate is present, then the next most energy producing reaction is manganese reduction, followed by iron reduction, and so forth, as listed in Figure 4-1A. As electron acceptors are consumed, more or less sequentially according to the energy released, the system becomes more reducing and has a lower pε. These naturally occurring sequential processes are shown in Figure 4-1B for an inundated soil. This figure schematically illustrates the range of redox processes that can occur in MUS systems during storage. For example, when water containing natural DOC is recharged for storage underground, there may be sufficient DOC to cause an aquifer that is otherwise oxidizing to become reducing. Alternatively, when oxygenated water is added to an aquifer, it can cause oxidation reactions.

There are a number of examples that demonstrate the importance of the redox potential to water quality and aquifer integrity in MUS systems. The persistence or degradation of many organic contaminants varies with redox potential. In general, organic compounds with greater halogen (Cl, Br, F) content are most readily degraded under reducing conditions and can serve as electron acceptors in the transformation process. In contrast, organic compounds that do not contain halogens (or have an insignificant amount), such as the aromatic hydrocarbons benzene and toluene, tend to be more readily transformed under more oxidizing conditions.[1] The importance of redox potential to contaminant persistence is illustrated in Box 4-3, which describes how different redox conditions during storage in MUS systems lead to variable formation and persistence of trihalomethane compounds (see Figures 4-2 and 4-3). When oxidizing water recharges an aquifer that contains reduced minerals, such as arsenopyrite (reduced iron sulfide) or other reduced forms of arsenic minerals, the minerals are oxidized and can release arsenic into the stored water. Conversely, when water containing DOC is recharged to an aquifer, reducing conditions that cause the release of iron and other metals and metalloids (including arsenic) into the groundwater can result in these constituents exceeding water quality criteria.

Changes in redox potential in an aquifer may have long-term consequences for aquifer integrity by enhancing either dissolution reactions (reactions that dissolve the aquifer media) or precipitation reactions that plug the aquifer. In addition to the redox reactions that directly dissolve or precipitate aquifer minerals, such as the pyrite oxidation described above that is also a mineral dissolution reaction, redox reactions have indirect consequences. For example, oxidation of organic matter creates acid products (partially transformed organic acids or carbonic acid) that chemically weather the aquifer media by dissolving the minerals in the aquifer. These reactions consume the acid and increase the dissolved salts and hardness of the stored water. Such reactions pose two potential issues for MUS. First, while the increase in dissolved salts may be relatively modest, changes in water quality may require treatment in some cases. Second, the impact of such reactions on water quality depends on the composition of the aquifer media. Knowledge of the aquifer mineral composition combined with geochemical modeling and/or standard bench-scale experiments may be sufficient to provide an initial assessment of the impacts of storage on water quality.

It must be emphasized that all of the above reactions are driven by the mixing of recharged water into an aquifer that creates conditions that are not in thermodynamic equilibrium.

[1] Note that determination of the redox state of the carbon in an organic compound allows one to determine the redox conditions under which a compound is mostly likely to be transformed, as described in Schwarzenbach, et al. (2003)..

BOX 4-3
Examples Demonstrating Contaminant Formation or Degradation with Differing Redox Conditions: Trihalomethanes

Disinfection by-products include several suites of primarily halogenated organic compounds that are formed when residual chlorine from disinfection reacts with natural organic matter. Trihalomethanes are often the predominant contaminant compounds formed as a consequence of these reactions. A few peer-reviewed publications and a much larger number of site reports have demonstrated that disinfection by-products, including THMs, can be formed in injection MUS systems where the injectate contains residual chlorine. However, these compounds can also be transformed during storage, and their persistence depends on redox conditions in the aquifer. The impact of different chemical conditions in the aquifer on contaminant fate is illustrated by the contrasting behavior of THMs in two aquifer storage and recovery (ASR) tests conducted with different redox conditions: conditions were dominantly aerobic during recharge and storage in the Yakima, Washington, pilot test, while anaerobic conditions (nitrate reducing to methanogenic) were present during the Bolivar test storage and recovery periods. Each of these pilot experiments is described below. A more detailed discussion of THM reactions and fate and transport can be found later in the discussion of disinfection by-products.

THM Formation and Persistence in an Aquifer: Yakima, Washington

The aquifer tested in this experiment is located in the Upper Ellensburg Formation, a geologic unit comprised of volcaniclastic sediments. The native groundwater is aerobic to microaerophilic as supported by the following water quality measurements: DO ~ 5 mg/L, 0.4 mg N/L nitrate, very low dissolved iron concentration (~0.018 mg/L), and no detectable manganese.

The experiment was conducted with recharge, recovery, and sampling from one ASR well and is described in Golder Associates Inc.(2001). The duration of the experiment was relatively short: water was recharged for 25 days, stored for 55 days, and recovered for 30 days. The total volume of water recharged was 1.7×10^5 m^3 (45.2 Mgal), and approximately twice as much water was extracted.

During this test, treated water from the Naches River, which is the primary municipal water supply for the City of Yakima, was used as recharge water. The water was disinfected using chlorine prior to recharge following the usual drinking water treatment method. Therefore, residual (unreacted) chlorine was present in the recharge water (~0.9 mg/L) along with a comparable amount of organic matter (total organic carbon content of recharge water was ~0.8 to 1 mg/L).

A relatively comprehensive suite of water quality parameters was measured at the ASR well during this test. In addition to the disinfection by-products, the following parameters were also monitored: major cations and anions; alumina and silica to allow interpretation of water-rock reactions; redox-sensitive species (such as iron and manganese); and environmental tracers, such as the stable isotopes ^{18}O and ^{2}H (deuterium) (described further below). Water quality samples were not collected from observation wells.

The environmental tracers ^{18}O and ^{2}H were present in distinctly different concentrations in the groundwater reservoir ($\delta^{18}O$ = −16.4 per thousand and $\delta^{2}H$ = −133 per thousand, both referenced to Vienna Standard Mean Ocean Water) compared to the recharged river water ($\delta^{18}O$ ~ −14.0 to -15.0 per thousand and $\delta^{2}H$ = −110 to −115 per thousand). No reactions occurred that markedly altered the tracer concentrations during the storage period. As shown in Figure 4-2B, the tracer concentrations in water extracted during the recovery period decline linearly from concentrations that indicate water was entirely recharged to concentrations that are comparable to native groundwater along a smooth "mixing" line (this indicates changing proportions with extraction volume).

Continues next page

BOX 4-3 Continued

The recharged water contained low concentrations of total THMs (TTHMs)(5-10 µg/L), comprising dominantly chloroform. During storage, TTHM concentration increased (Figure 4-2C) when residual chlorine residual was present (Figure 4-2B), indicating formation of disinfection by-products in situ, presumably as the residual chlorine reacted with the injected organic matter. Increases in THM concentrations did not occur following depletion of the chlorine residual. TTHM concentrations declined linearly during the recovery phase, following a trend similar to the environmental tracers. This trend suggests that the concentrations declined primarily because the THMs were flushed from the aquifer rather than because they were transformed. It is perhaps noteworthy that the TTHM, chloroform, and dichlorobromomethane concentrations observed in the aquifer during the storage phase were below the existing drinking water standard concentrations for these compounds. However, the latter two compounds exceeded the groundwater quality standard for Washington State. This experiment demonstrated geochemical conditions in which contaminants of concern were formed in the aquifer during storage and were persistent for the duration of this short experiment. (Additional detail about this pilot test is reported in Golder Associates Inc., 2001.)

THM Attenuation Associated with Reducing Conditions

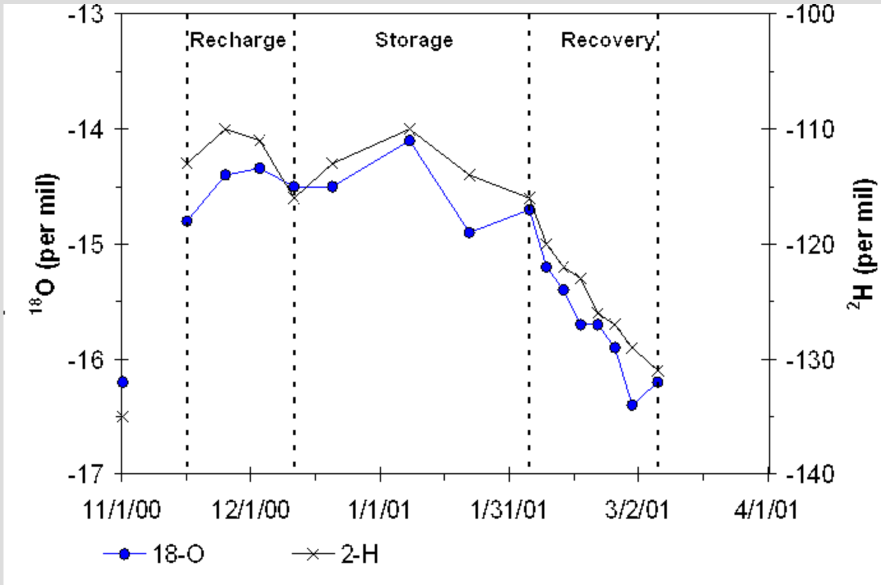
In contrast to the above experiment, a field ASR trial conducted in an anoxic aquifer at the Bolivar site, near Adelaide, Australia, demonstrated that THMs can be significantly attenuated during storage. The aquifer comprises marine-deposited limestone, and the native groundwater has very low dissolved oxygen (0.1 mg/L) and a redox potential of 42 mV.

In this experiment, a total of ~2.5×10^5 m^3 (similar to the Yakima experiment) of chlorinated reclaimed water was recharged, about 85 percent of this was injected at a relatively continuous rate over eight months. The storage period (~3.5 months) was nearly twice as long as that of the Yakima experiment. The volume extracted was equivalent to approximately 60 percent of the volume injected. The recharged water contained residual chlorine (~0.7 mg/L) along with a much greater organic carbon concentration (average = 18.2 mg/L) than Yakima. This experiment also included comprehensive water quality sampling and analysis (redox indicators, dissolved nutrients, and tracers in addition to contaminants) from several monitoring wells and the ASR well. Chloride ion served as the conservative tracer in this experiment because the recharged water and native groundwater have distinct concentration ranges (recharged water = 430 $\pm$ 40 mg/L and ambient groundwater = 930 $\pm$ 90 mg/L).

THM formation occurred in the aquifer during recharge when chlorine residual was present. While the THM concentration measured at the 4-m (from the ASR well) observation well was as high as ~140 µg/L, the concentrations declined rapidly during storage (Figure 4-3A and 4-3B), while the chloride tracer concentrations remained constant (behavior consistent with degradation reactions). It was also noted that the THM attenuation occurred more rapidly in the ASR than in the observation well. The difference was consistent with a difference in redox potential in these two locations. More rapid attenuation at the ASR well compared to the observation well was attributed to the more reducing conditions that developed in the ASR well during storage (methanogenic conditions were observed at the ASR well and nitrate reducing conditions were observed at the observation well). The authors of this study point out that although DBPs were formed during the first week of aquifer storage, their long-term behavior was controlled by degradation reactions that are reasonably fast compared to typical storage cycles. (Additional detail about the Bolivar field trial is available in Pavelic, 2005c.)

This Bolivar field trial and the contrasting Yakima pilot experiment demonstrate how differences in geochemical conditions, in this case redox potential, either native to the

groundwater system or conditions that develop during recharge and storage can control both the rate and the extent of contaminant formation and attenuation processes.



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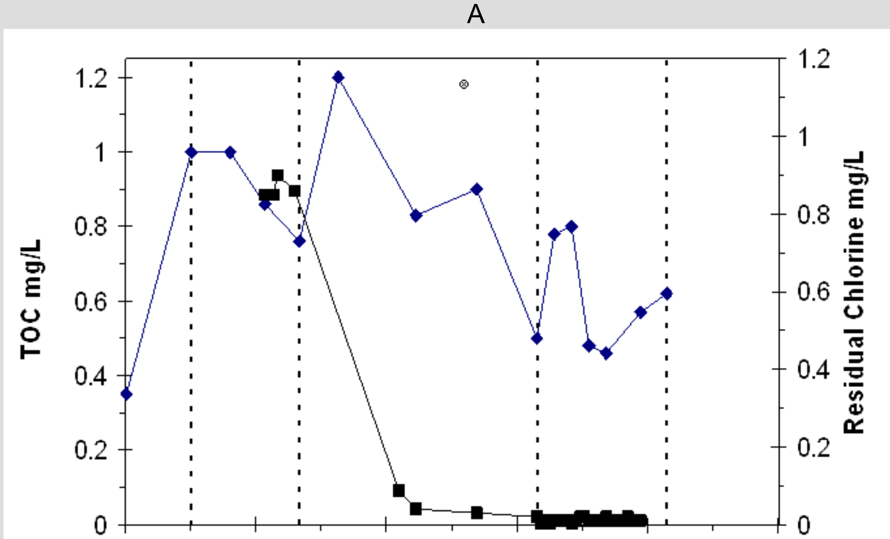

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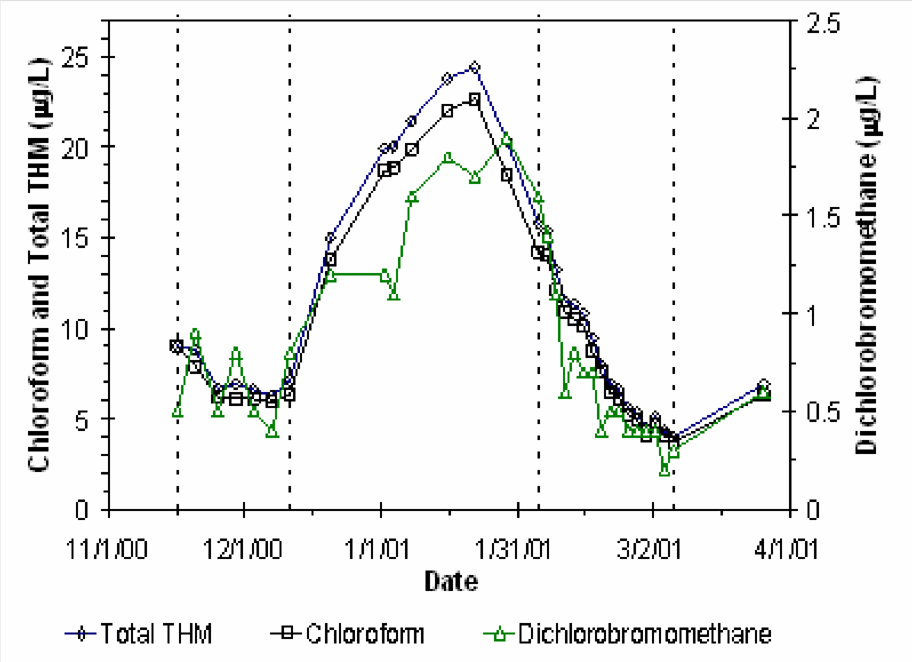
BOX 4-3 Continued



C

FIGURE 4-2 (A) Environmental tracers, (B) TOC and residual chlorine, and (C) selected and total THMs in the Yakima, Wash., ASR pilot study. The pilot test is described in the main text Box 4-3. After Golder Associates Inc. (2001). Reprinted, with permission, from Golder Associates (2001). Copyright 2001 by Golder Associates.

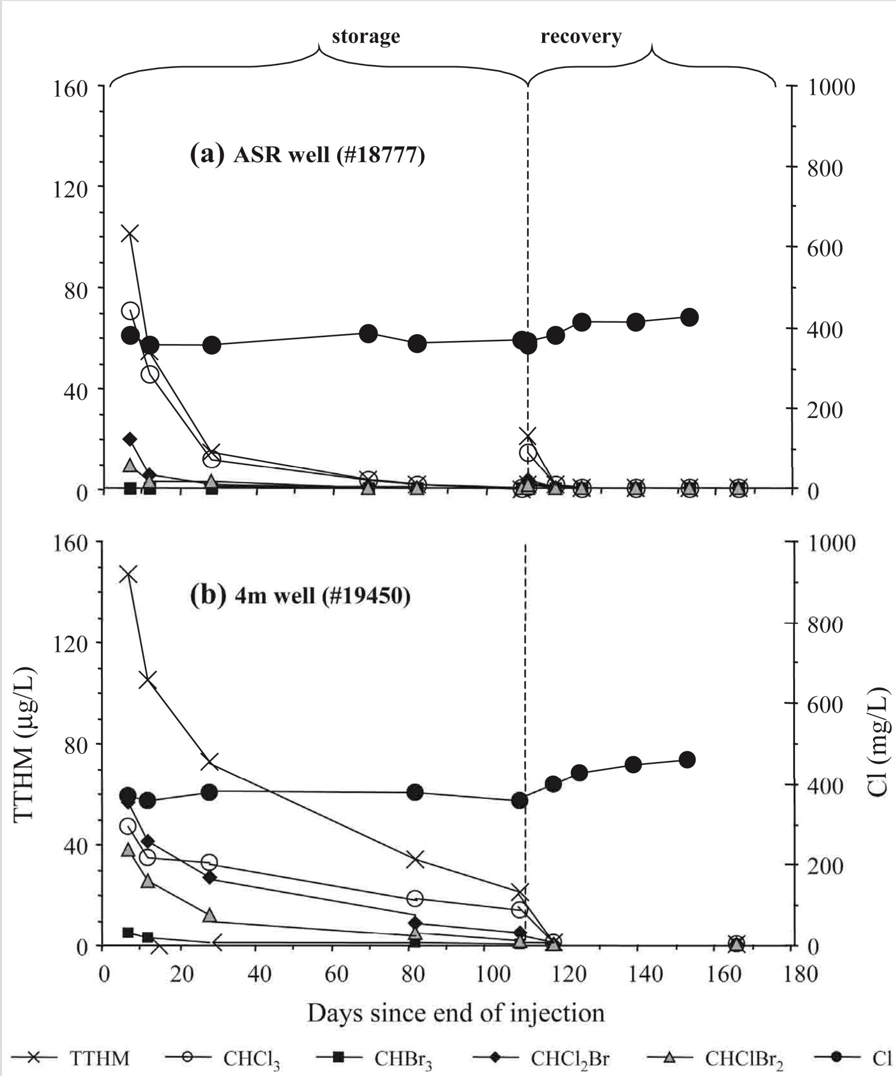


FIGURE 4-3 Declining concentrations of individual and total THMs during storage as measured in the recovery and observation wells compared to the near constant (conservative) concentration of the chloride ion (Cl^-) tracer indicate that the THMs were transformed. Removal of these halogenated compounds was more rapid (shorter half-lives) in the ASR well (a) where the water has the lowest redox potential (methane production observed) compared to the observation well, and (b) where nitrate reducing conditions were observed. Figure taken from Pavelic et al. (2005 b). Reprinted, with permission, from Pavelic (2005). Copyright 2005 by Elsevier Limited.

Precipitation-Dissolution Reactions

Precipitation and dissolution reactions are driven by thermodynamic disequilibrium between the dissolved ions and the mineral (solid) phase(s). These reactions are important to aquifer integrity because they can result in either increased or decreased aquifer porosity and permeability. The formation of karst landscapes, including caves and sink holes, is a natural example of aquifer dissolution over a very long time. In MUS systems that cause mineral dissolution reactions, the MUS system accelerates chemical weathering processes in the aquifer and may also alter the spatial distribution of such reactions compared to the natural situation. Furthermore, it should not be assumed that dissolution reactions occur uniformly in either space or time, but they will occur most readily in preferential zones. Precipitation and dissolution reactions can also significantly affect water quality and can exert a particularly important impact on the concentrations of some regulated metals and metalloids (described later in this chapter).

Dissolution reactions are favored when recharged waters contain relatively low dissolved ion concentrations compared to the native groundwater, which is frequently the case for surface water sources. Such waters are likely undersaturated relative to the aquifer minerals resulting in dissolution reactions that trend towards the equilibrium condition. For example, the major minerals of a limestone aquifer, comprised of calcite ($CaCO_3$) and/or dolomite ($CaMg(CO_3)_2$) in significant proportion, would be dissolved readily by waters that are either acidic or contain low base cation (Ca and Mg in this case) and bicarbonate concentrations. Dissolution reactions can occur in aquifers comprised of silicate minerals wherein the silicates are also dissolved (chemically weathered) by acidic waters. However, the mass of aquifer solid removed from dissolution of these types of rocks by each pore volume of fluid exchanged is typically small compared to the limestone case. The reasons are twofold: silicate mineral solubility is relatively low compared to the solubility of carbonates at circumneutral pH and kinetic constraints may limit dissolution reactions. Therefore, in silicate rocks, the dissolution process of removing the aquifer matrix tends to be much more damped compared to the behavior of limestones.

Microbiological activity may also play a role in precipitation and dissolution. For example, iron reducing bacteria can cause reductive dissolution of Fe(III) (hydr)oxides in the presence of labile organic carbon. This process releases Fe(II) and other metals associated with the solid phases (e.g., arsenic and nickel) into the water. This process has much less effect on aquifer integrity than it does on water quality.

A more complex example of dissolution in carbonate aquifers can arise because carbonate mineral solubility is a strong function of environmental conditions (e.g., temperature and carbon dioxide partial pressure). Prior to recharge and mixing due to MUS activities, water in the aquifer is likely in equilibrium with its matrix carbonate minerals. Upon mixing with recharge or source water, however, the “new” water may be undersaturated with respect to the host aquifer minerals because of a difference in environmental conditions. The mixed water is therefore chemically aggressive. To reestablish equilibrium, the aggressive water will dissolve calcite and dolomite, which results in changes in water composition. Although only a small proportion of the total aquifer solids is removed by this process by each volume of water to which it is exposed, such reactions could eventually affect aquifer integrity. Moreover, preferential flow paths in the aquifer (see Chapter 3 section on dual porosity) may also develop if the process were to continue for a long period of time. In preexisting dual-porosity storage zones, rapid water quality changes may occur due to mixing in conduits and fractures. Stuyfzand (1998) and Herczeg and colleagues (2004) are among the researchers that provide more detail on the effects of these geochemical processes during aquifer storage and recovery (ASR) activities. In addition, the behavior of selected contaminations in response to precipitation and dissolution is discussed later in this chapter.

Precipitation reactions in MUS systems are most likely to occur as a consequence of redox changes. Sulfate reduction, for example, produces hydrogen sulfide and bicarbonate. Under reducing conditions (low ORP or Eh), dissolved sulfide and iron precipitate as reduced iron sulfide minerals that incorporate metal cations (such as zinc and nickel), thus reducing the concentrations of these metals in solution. Another example is that of reduced iron containing water experiencing an increase in Eh (as it would during extraction) leading to precipitation of ferrous iron (hydr)oxides. In addition to potentially clogging the aquifer, ferrous hydr(oxides) are excellent metal sorbents. They “scavenge” arsenic and other metals from the dissolved phase through coprecipitation and sorption, thus reducing the dissolved concentrations of the scavenged elements.

Sorption of Organic Compounds

Sorption is the term used to describe the transfer of a chemical from the aqueous to the solid phase without reference to the mechanism of the compound-solid interaction. Desorption is the reverse process. The solid phase may be an inorganic or organic constituent. With regard to inorganics, a change in hydrochemical conditions in the aquifer due to MUS activities may cause desorption of metals that are weakly bonded to minerals comprising the aquifer matrix. Sorption-desorption processes are complex and perhaps are best illustrated in the context of organic compounds.

For most low-polarity and apolar organic contaminants—such as many on the regulated chemical list (Appendix A, Table A-1)— the primary sorbent is the natural carbonaceous matter (noncarbonated, carbon-containing material such as humic substances, char, and kerogen) in the aquifer. For these chemical-sorbent combinations, sorption is generally reversible and the forces binding the contaminants to the sorbent are relatively weak (van der Waal forces). In addition to these forces, ionizable and polar organic compounds can also interact with the mineral surfaces of the aquifer solids through dipole and electron donor-acceptor interactions. These interactions also generally contribute to reversible sorption. Extensive discussion of the thermodynamics underlying organic compound sorption, as well as the effects of variable compound and aquifer solid properties on the magnitude of sorption, are provided in several texts and reviews, including Allen-King et al. (2002); Cornelissen et al. (2004); and Schwarzenbach et al. (2003).

Reversible sorption (or desorption) acts as a temporary storage reservoir for contaminants in the aquifer. Once the aquifer solids equilibrate with a particular dissolved contaminant concentration, the sorption-desorption process will not have any further net effect on dissolved concentration. For example, Miller et al. (1993) found that THMs were not appreciably affected by sorption during a field storage and recovery operation in Las Vegas. In the context of an MUS system, reversible sorption-desorption will cause the velocity of organic contaminant transport to be retarded compared to the water velocity when water is added to storage. Over short time scales (prior to equilibration), sorption will attenuate dissolved concentrations. If the source contains a variable concentration of a contaminant, sorption-desorption processes during transport and storage may serve to damp the variability in the dissolved concentration of water extracted from the MUS system. Therefore, reversible sorption does not provide a sustainable contaminant sink because the compounds are not removed from the MUS system (as they are when contaminants are biodegraded, for example).

Although the forces causing sorption are not particularly strong, the mass taken up by the solid phase can be significant. The magnitude of the sorption process and its dependence on concentration are functions of the specific physicochemical properties of the carbonaceous matter and organic contaminant. Sorption can be nonlinear in concentration, and co-solutes may compete for more energetically favorable sorption sites, particularly when compounds are present at low concentrations compared to contaminant solubility.

The effects of sorption-desorption may be more apparent and of greater impact on contaminant recovery during short-duration or small-scale tests (lab and pilot-scale studies) than in full-scale operations. In such tests, the source water and aquifer solids may remain farther from equilibrium than they would be during full-scale operations. Therefore, such tests must be conducted and interpreted such that extrapolation to a longer-duration and larger-scale system appropriately accounts for sorption/desorption dynamics.

Ion Exchange Reactions

Another water-rock interaction process that can occur during MUS activities is cation exchange. Positively charged ions with physical and chemical affinities can be exchanged between the water and the minerals comprising the aquifer matrix. Common examples involve the exchange between Ca^{2+} or Mg^{2+} with Na^{+} or K^{+}. Mineral groups primarily involved in these reactions are clays and zeolites because of their relatively high surface areas compared to others. In the case of clay minerals, for example, K^+ in the clay may exchange with Ca^{2+} in the water. This process does not change the total amount of charged species dissolved in the water. However, it can cause significant changes in the concentrations of various ions dissolved in the water. As the aquifer is repeatedly exposed to the recharged water, the composition of exchangeable ions associated with the aquifer solids will change, evolving toward quasi equilibrium with the recharged water. This process can also significantly affect the dissolved concentrations of trace metal cations.

Particle and Microorganism Transport

The movement and fate of particles and microorganisms that may be in source waters for MUS systems is of interest. Particle composition can include organic matter that can support redox reactions, pathogenic or innocuous microorganisms, minerals, and aggregates of any combination of these. In addition, several classes of contaminants, such as hydrophobic organics and certain toxic metals, associate with particles. Their movement in the subsurface is influenced by the behavior of the particles, not only by the dissolved phase concentrations. If the extracted water is used for drinking, then effective particle capture is desired so that the turbidity falls below the drinking water standard. Microorganism transport and survival in MUS systems is especially important when the microorganisms are pathogenic. The subsurface can be an effective sink for removing pathogens to improve the quality of the extracted water. Finally, the movement of microorganisms and particulate organic matter influences the distribution of microbial activity within an MUS system. This in turn will impact the spatial distribution of microbial activity in the storage zone and the extent and rates of biotransformation reactions.

The typical grain sizes that exist in the subsurface and the associated moderate to high specific surface area means that effective filtration and particle removal is often possible in MUS systems. The capture and accumulation of microorganisms on surfaces often enhances the potential for biotransformations. Particle and microorganism transport is typically governed by movement of the groundwater coupled with retardation by attachment onto surfaces and straining or trapping in interstitial pores. Attachment is commonly thought of as the main contribution to retardation and removal. Removal by straining is thought to be important only when the diameter of the particle exceeds 5 percent of the mean interstitial pore size (Jenkins and Lion, 1993; McDowell-Boyer et al., 1986). Particle and microorganism transport through the subsurface is influenced by several parameters including properties of the particle and microorganism, solution chemistry, subsurface media characteristics, and interstitial fluid velocity. These factors are briefly described in the following paragraphs. Several reviews of particle and microorganism behavior in porous media are available if the reader desires additional information (McDowell-Boyer et al., 1986; Bouwer et al., 2000; MWH, 2005; Tufenkji, 2007).

Particle and microorganism size and shape as well as surface charge and hydrophobicity influence transport, retardation, and adhesion to surfaces. The presence of molecules such as proteins or polysaccharides on the cell surface and the presence of pili, as well as motility and chemotaxis, influence microorganism behavior in porous media. Many cell properties are influenced by the physiological state of the microorganism and can therefore differ significantly for the same species depending on environmental conditions. The growth state of the microorganism and the presence of nutrients have, for instance, been shown to influence attachment (Cunningham et al., 2007). Starvation is another important physiological state of microorganisms. Short-term starvation of bacteria can result in an increased tendency to attach to surfaces. Long-term starvation (weeks to months) in contrast may enhance microbial transport through porous media.

Solute characteristics including ionic strength, pH, temperature, concentrations of dissolved organic matter, surfactants, and nutrients have also been shown to influence particle and microorganism transport and adhesion to surfaces. Increased ionic strength has been correlated widely with increased attachment. This effect is usually attributed to the compression of the electrostatic double layer in the presence of high ion concentrations. Changes in solution pH have been shown to either increase or decrease the extent of particle and microorganism transport and attachment. Consequently, uniform results for the influence of pH have not been observed. Dissolved and sediment organic matter has been shown to increase the travel distance for particles and microorganisms in porous media columns. The addition of surfactants or dispersants can result in decreased attachment and therefore facilitate the transport of particles and microorganisms through porous media; however the activity or viability of the microorganisms may also be altered.

Porous media properties that have been reported to influence particle and microorganism transport and adhesion include pore water velocity, hydraulic conductivity, pore size, surface roughness, the presence of iron minerals and other surface coatings, the organic matter content, and grain and pore size distribution. The surface charge and surface hydrophobicity of the porous media can also influence particle and microorganism attachment to surfaces.

Transport of particles and microorganisms through porous media may be influenced by some combination of the foregoing parameters. Measurements of particle and microorganism attachment and movement under the conditions of interest tend to be much better predictors of movement and fate than attempting to scale-up information from characterization of the particles or cells or the porous medium. One approach to predicting particle and microorganism transport through porous media is to perform experiments with the aquifer material of interest as close as possible to the expected conditions in the field. Harvey (1997) provides a good overview on how to design and standardize bacterial transport experiments.

Microbial Inactivation

Inactivation or death is an important mechanism that causes the removal of microorganisms from recharged water during storage in MUS systems. Attenuation of microbial contaminants of concern, including viruses and parasites, in surface, groundwater and MUS systems has focused on understanding the survival kinetics influenced by environmental conditions. It is known that the inactivation rates can be described by the following:

- Type of microbe. Parasites and viruses are more resistant then bacteria; however, bacteria (particularly coliforms) may regrow at higher temperatures.
- Temperature. Increased temperature typically increases the activity of native microbes and also directly influences inactivation rates of nonnative microbes, with higher temperatures leading to greater inactivation rates; for example, between 10 and 200 days are needed to achieve 99 percent inactivation of *Cryptosporidium* depending on the temperature (Table 4-3).
- Redox potential. Greater survival has been reported under anaerobic conditions in several studies.
- Native microflora. Influenced by temperature, nutrients, and aerobic conditions, increased activity generally enhances inactivation rates of fecal organisms.

Enteric microorganisms of wastewater origin have been the predominant focus of studies on survival in groundwater with temperature the predominant variable studied. A recent review by John and Rose (2005) examined all reports describing microbial inactivation in groundwater and summarized inactivation rates for bacteria and viruses. The analysis showed that only temperature and type of microorganism influenced the inactivation rate (Figure 4-4). The data represented a mixture of studies done under aerobic and anaerobic, sterile and nonsterile conditions, but often there were not enough studies with the same organism under the same temperature to show a statistical difference. Nonsterile conditions more often showed a greater inactivation than did sterile conditions when contrasted.

Rates of decline for fecal coliform bacteria in the literature were highly varied at 5 ºC (ranging from an inactivation of -0.02 to -0.14 $\log_{10}\ d^{-1}$) with the geometric mean of summarized coliform inactivation rates for temperatures less than 10ºC equal to −0.05 $\log_{10} d^{-1}$. At higher temperatures (21 - 37ºC) coliform inactivation averaged about −0.1 $\log_{10} d^{-1}$ (geometric mean). This may indicate that regrowth is contributing to the overall inactivation rates. Similarly, regrowth of *Enterococci* may be occurring in groundwaters at higher temperatures, reflected in an overall slower inactivation rate. Pathogens such as *Salmonella* show an increasing rate of inactivation with increasing temperature, where as others such as *Shigella* exhibit variable rates and reflect the differences in

TABLE 4-3 First-Order Inactivation Rates of *C. parvum* in Natural Water Samples at Three Temperatures

Water Type	Water Source	Temperature (°C)	Linear Inactivation Rate ($log_{10}d^{-1}$)	Estimated Days to 99% Decline	Standard Deviation
Groundwater	Avon	5	0.0088	>200	
	Park Aquifer	22	−0.0010	>200	
		30	−0.11	18	
Groundwater	Lake	5	0.00090	>200	
	Lytal Aquifer	22	−0.042	48	
		30	−0.12	17	
Surface water	Bill Evers	5	−0.0017	>200	
	Reservoir	22	−0.045	45	
		30	−0.20	10	
Surface water	Clear Lake	5	−0.0037	>200	
	Reservoir	22	−0.0066	30	
		30	−0.18	11	
Groundwater	Avon	5		>200	0
	Park andLake	22		124	107
	Lytal Aquifer	30		17.5	0.71
Surface water	Bill Evers	5		>200	0
	and Clear	22		37.5	10.6
	Lake	30		10.5	0.71
	Reservoir				

SOURCE: Ives et al. (2007). Reprinted with permission from Ives et al. (2007). Copyright 2007 by American Society for Microbiology.

experimental design associated with aerobic or anaerobic conditions and microflora background.

It is known that viruses do not regrow in the environment, and inactivation rates in the virus literature show a clear temperature affect. Inactivation rates of coliphage (a fecal bacterial virus indicator) in groundwater were also summarized by John and Rose (2005). Below 10ºC the geometric mean rate was -0.03 log_{10} d^{-1}, however, at a moderately high temperature range of 21 -25 ºC, the summarized coliphage inactivation rates increased tenfold averaging −0.3 log_{10} d^{-1} (geometric mean). Enteric viruses were very stable (−0.02 log_{10} d^{-1}) below 21°C. Some viruses (e.g., hepatitis A virus) were stable at all temperatures.

Another potential factor controlling the fate of fecal microorganisms, both in groundwater and in surface water, is the activity of other microorganisms such as bacteriophages, bacterivorous protozoa, and antagonistic autochthonous bacteria. While some studies have demonstrated that the presence of native bacteria increased inactivation of seeded organisms (Banning et al., 2002; Gordon and Toze, 2003; Janakiraman and Leff, 1999; Kersters et al., 1996; Medema et al., 1997; Sobsey et al., 1986) others have shown inconsistent effects (Yates and

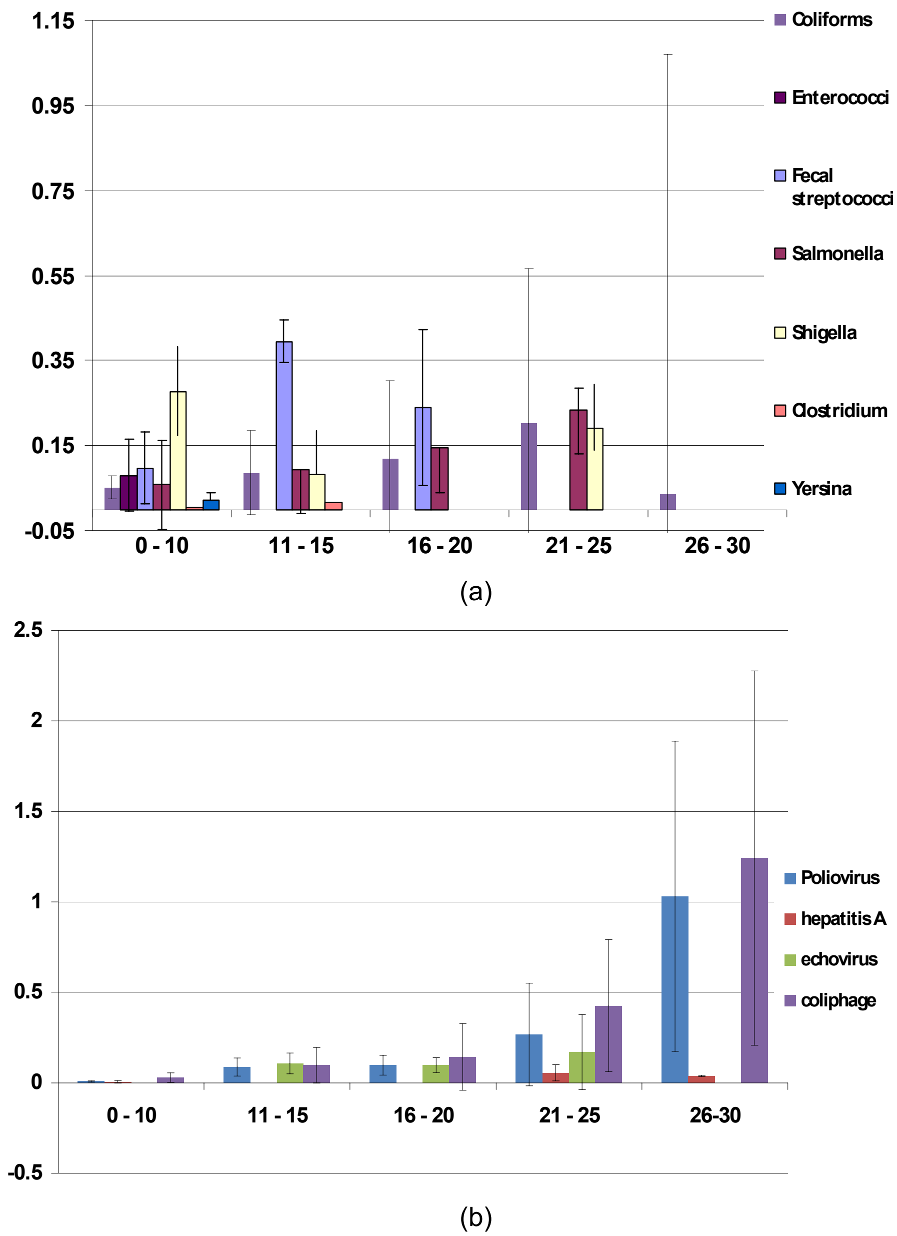


FIGURE 4-4 Mean inactivation rates of bacteria (a) and viruses (b) in groundwater by organism type and temperature. Values review by John and Rose (2005). Error bars refer to one standard deviation in $\log_{10}$ per day. Adapted from John and Rose (2005). Copyright 2005 by American Chemical Society.

Gerba, 1985; Yates et al., 1990) or the opposite (Alvarez et al., 2000). This may have to do with interaction between oxygen levels, temperature, and nutrients. Studies undertaken using *Escherichia coli* showed inactivation rates of −0.11 $\log_{10} d^{-1}$ followed by increased rates of -0.35 $\log_{10} d^{-1}$ under aerobic conditions, while under anaerobic conditions the inactivation was -0.02 $\log_{10} d^{-1}$ (Roslev et al., 2004). Gordon and Toze (2003) showed that microbial flora in groundwater influenced by oxygen, nutrients, and temperatures influenced survival rates of enteric viruses.

Appendix A discusses some of the specific pathogens of concern. Some bacteria are able to regrow, which include the indicator bacteria *Arcobacter* and *Legionella*, yet models that can predict regrowth in the water environment are not available as they are for food. Parasites and viruses do not regrow but will survive. There is a need to undertake further research to describe the inactivation rates. Tracer studies with septic tanks show that long-term viral contamination of the soil drain fields with pulses released associated with rainfall events (Nicosia et al., 2001). While initial inactivation may be rapid, often the data show long-term tailing effects that have not been well described.

Numerous reports have also suggested that attachment to mineral surfaces reduces viral inactivation rates (Rossi and Aragno, 1999; Ryan et al., 2002; Sakoda et al., 1997) and stream sediments likely confer protection on fecal bacteria from inactivation in surface water (Buckley et al., 1998; Crabill et al., 1999; Sherer et al., 1992). However, studies on MS-2 and PRD-1 bacteriophage (Blanc and Nasser, 1996) and *Enterococcus faecalis* (Pavelic et al., 1998) have shown more rapid inactivation in water with solid media present or no difference.

Biotransformations

The metabolic capabilities of subsurface microorganisms are quite diverse. For growth of microorganisms, electron donors and acceptors, a carbon source, and essential nutrients are required. Either natural or anthropogenic compounds in source or native groundwaters can provide these growth requirements in MUS systems. Chemicals that are electron donors are oxidized during microbial metabolism to yield energy for growth. Oxidation can take place aerobically (in the presence of oxygen) or anaerobically (in the absence of oxygen). When molecular oxygen is available, it is generally the preferred terminal electron acceptor of electrons that are released during the oxidation of electron donors. As an electron acceptor, oxygen can be replaced by other oxidized inorganic compounds, such as nitrate, metal ions (e.g., Fe(III), Mn (III), or Mn(IV)), sulfate, or carbon dioxide, although the energy gains to the microorganisms are then smaller. These alternate electron acceptors are reactants in anaerobic microbial processes. Microbial reactions in MUS systems can contribute to changes in redox conditions within the storage zones. These redox changes in turn can influence the water quality in MUS systems.

Biotransformations of chemicals in the storage zone offer the prospect of improving water quality during MUS. Many classes of organic compounds, such as natural organic matter, petroleum compounds, halogenated compounds, some pesticides, and endocrine disrupting compounds, are known to be biotransformed by subsurface microorganisms. In some instances, the compounds are the primary energy and carbon supply for microorganisms. For other compounds, the biotransformation occurs as cosubstrate utilization where enzymes involved in the metabolism of one substrate are also able to degrade the contaminant. Several reviews cover the topic of subsurface contaminant biotransformation (Atlas and Philip, 2005; NRC, 1993; 2000; Young and Cerniglia, 1995;). Examples of compound biotransformations that have been observed in MUS systems are described elsewhere in this chapter.

BEHAVIOR OF SELECTED CONTAMINANTS IN MUS SYSTEMS

Empirical and experimental evidence from established MUS systems demonstrates that water quality objectives can be met consistently by underground storage systems over long periods of time—decades. In some of these systems, especially those that use recharge basins, subsurface treatment removes a portion of the contaminants in the source water, thus the subsurface recharge and storage system plays an integral role in improving water quality.

The following sections draw on available field and laboratory studies to describe the processes that affect the behavior of several contaminant classes that are of particular importance to MUS systems. The contaminants described were selected because they either strongly affect operations (e.g., dissolved organic carbon); are regulated contaminants; are among the more frequently detected or persistent contaminants of concern in MUS; or warrant additional consideration in such storage systems because of lack of complete information.

Organics

This section focuses on only three groups of organic constituents that are particularly important to MUS systems: total organic carbon (TOC), disinfection by-products, and pharmaceuticals and personal care products (PPCPs) and other emerging contaminants of concern. These compounds either are frequently detected in the source waters used for MUS or can be created by in situ subsurface reactions. The fates of other classes of anthropogenic organic chemicals in groundwater, such as chlorinated solvents, regulated pesticides, and petroleum hydrocarbons, must also be considered for any particular MUS if these compounds are present in either the source water or the groundwater system. The fate of these contaminants in groundwater is relatively well documented by research and literature on groundwater remediation of point source spills of chlorinated solvents, hydrocarbons, and other industrial chemicals and by similar work on agricultural pesticides that can occur in groundwater through either point or nonpoint discharges. (NRC, 2002, 2004).

Organic Carbon

Organic compounds are removed during subsurface storage by a combination of filtration, sorption, oxidation-reduction, and biodegradation. Biodegradation is the primary -sustainable removal mechanism for organic compounds during subsurface transport. DOC can lead to the formation of DBPs upon addition of a disinfectant. Furthermore, the degradation of labile DOC and particulate organic carbon in recharge water can also cause clogging because it promotes high biomass growth. This topic is addressed elsewhere in the chapter.

The concentrations of natural organic matter and soluble microbial products (SMPs) that comprise the bulk of the dissolved and particulate organic carbon are reduced during subsurface transport as high-molecular-weight compounds are hydrolyzed to lower-molecular-weight compounds and the lower-molecular-weight compounds serve as substrates for microorganisms. As the concentrations of NOM and SMPs decrease, the disinfection by-product potential associated with these compounds also decreases (AwwaRF, 2001). In addition, synthetic organic compounds at concentrations too low to directly support microbial growth may be co-metabolized as NOM and SMPs serve as the primary substrate for growth. Given sufficient surface area and contact time, the water used for underground storage may approach the quality of native groundwater with respect to DOC concentration.

The transformation of organic compounds during recharge may be divided functionally into two regimes defined as short-term transformations, wherein relatively fast reactions occur, and long-term transformations, wherein recalcitrant compounds transform at slower rates over time. Short-term transformations occur in less than ~30 days and consume the majority of easily biodegradable carbon. The easily biodegradable carbon can be assessed by the biodegradable dissolved organic carbon test (BDOC). Box 4-4 and Figure 4-5 illustrate both the reduction and the change in composition of DOC in reclaimed water that can occur during recharge using surface spreading.

Disinfection By-Products

Disinfection by-products (DBPs) are formed as a consequence of reactions between disinfection chemicals (chlorine and chloramine) used to treat microbial pathogen contaminants and DOC. They are often small, halogenated (e.g., chlorinated, brominated) or nitrogen-containing organic compounds. Because the precursor NOM is complex and of variable composition, the DBPs produced encompass a spectrum of chemicals including the regulated trihalomethanes and haloacetic acids, and emerging nitrogen and halogen-containing contaminants

BOX 4-4
DOC Reduction and Change in Composition
During Recharge at the Mesa Northwest Water Reclamation Plant (NWWRP)

Figure 4-5(a) presents data with nitrified-denitrified reclaimed water at the Mesa NWWRP that illustrate short-term transformation of DOC. This study compared a field recharge basin site with soil column studies completed under aerobic and anoxic conditions. After 20 days, the final DOC concentration was similar under all conditions (Fox et al., 2001). Under aerobic conditions, the majority of easily biodegradable DOC was removed after several days, while 20 days were required for comparable removal in the anoxic column experiment. Since the time scales used for most groundwater recharge systems might be on the order of months, the removal of BDOC under aerobic conditions or anoxic conditions was similar. The NOM of the drinking water source for these experiments was approximately 2 mg/L, while the persistent SMPs contributed by wastewater treatment amounted to approximately 1 mg/L for a total of 3 mg/L DOC after short-term soil-aquifer treatment during recharge.

As water passed through the saturated time zone over longer time scales, long-term transformations of organic carbon continued. These transformations were similar to those that occurred when the natural recharge of surface waters into aquifers resulted in water quality improvements. The DOC concentration as a function of distance at the groundwater recharge basins is presented in Figure 4-5(b). The recharged reclaimed water was anoxic. Each 1,000 feet of travel was equivalent to approximately 6 months of travel time. At the monitoring wells closest to the basin, the DOC concentration was reduced to a concentration lower than the original drinking water DOC concentration. After several years of travel time, the DOC concentrations were less than 1 mg/L as they approached the background concentrations of the aquifer.

At this field site, the organic matter was also characterized in detail to allow comparison between the DOC composition and structure in the final product of a groundwater recharge system and the NOM present in the original drinking water source, Samples representative of reclaimed water before groundwater recharge, after short-term subsurface transformations, and after long-term subsurface transformations were analyzed. Spectroscopic characterization by ^{13}C-nuclear magnetic resonance and Fourier transform infrared did not find any significant differences in the major functional groups (AwwaRF, 2001). Major differences were identified in the organic nitrogen content in the reclaimed water (treated wastewater) compared to NOM because of the contribution of SMPs. This difference was also verified by fluorescence spectroscopy. However, after long-term subsurface transformations, the elemental composition and fluorescence of the groundwater recharge product water resembled NOM. The majority of differences between reclaimed water organic matter and NOM were eliminated by short-term transformations. Based on state-of-the-art techniques used to characterize the DOC, the bulk organic matter in groundwater recharge product water could not be distinguished from NOM.

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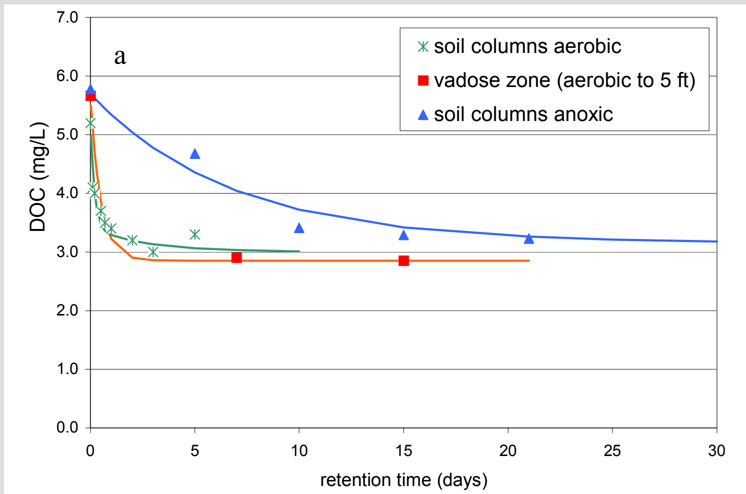
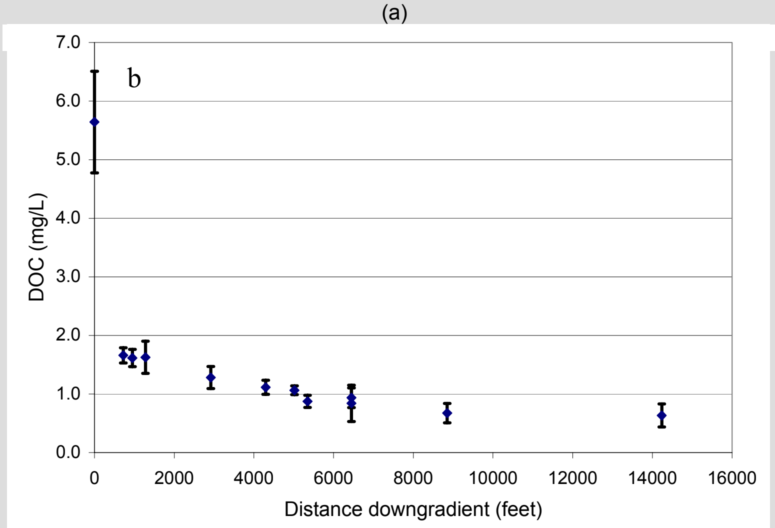
BOX 4-4 Continued



(a)



(b)

FIGURE 4-5 Dissolved organic carbon concentrations at the Mesa Northwest Water Reclamation Plant as a function of (a) retention time in the vadose zone and in aerobic and anaerobic laboratory column experiments, and (b) travel distance in the groundwater.

such as *N*-nitrosodimethylamine (NDMA), 1,1,1-trichloropropanone, and trichloroacetonitrile. Nitrogen-containing DBPs are a more frequent product of chloramination. Emerging DPBs are discussed in the next section.

Excess (residual) disinfectant is purposefully added to drinking water prior to transmission through distribution systems and may be added prior to recharge to control biological activity during recharge (Fox et al., 1998). Under these conditions of available NOM and residual disinfectant, DBPs are formed during aquifer storage (Pavelic et al., 2006; Thomas et al., 2000), as well as in the treatment process. Because the DBP formation potential is affected by the concentration and composition of the DOC, DBP formation varies both among aquifer storage systems and with source water quality changes for a particular storage system (Pavelic et al., 2006).

Transformation is the primary process that reduces DBP concentrations during aquifer storage. Literature supporting this point includes both field studies of storage systems and laboratory studies that determine the conditions under which transformation occurs, the rates and mechanisms of transformation, and the products formed. The reactivity of the DBP (or DBP subgroup, such as THMs) as well as the geochemical conditions of storage also affect the rate, mechanism, and product distribution. Because most DBPs are small, relatively soluble organic molecules, retardation due to sorption in at least low carbon content aquifers is relatively limited.

Of the various groups of DBPs, the greatest amount of research into persistence is available for the THMs. Field and laboratory studies demonstrate that THM persistence depends strongly on geochemical conditions, including redox state and electron acceptor and donor availability, as well as the compound considered. THMs are transformed in reducing (anaerobic) systems by reductive dehalogenation and persistent in aerobic systems limited in organic matter. In their recent manuscript, Pavelic et al. (2006) compared estimates of total THM persistence (in terms of half-life) during storage at eight different aquifer storage sites that represent a range of geochemical conditions. They showed that THMs are much more persistent in storage zones that remain aerobic (Figure 4-6) compared to anaerobic systems. Further, the attenuation rates reported vary by more than two orders of magnitude.

Chloroform frequently comprises a significant or dominant fraction of the total THM concentration present in stored water (Pavelic et al., 2006). Several studies have shown that chloroform is resistant to transformation or persistent in nitrate reducing biotic systems and iron reducing abiotic and biotic systems (Chun et al., 2005; Landmeyer et al., 2000; Niemet and Semprini, 2005). However, chloroform biotransformation is well known for methanogenic conditions in both field aquifer storage experiments and laboratory studies (Bouwer and McCarty, 1983; Pavelic et al., 2005).

The first-order transformation rates reported for brominated THMs are greater than those of chlorinated compounds for identical redox conditions (Kenneke and Weber, 2003). This finding supports the field observation that

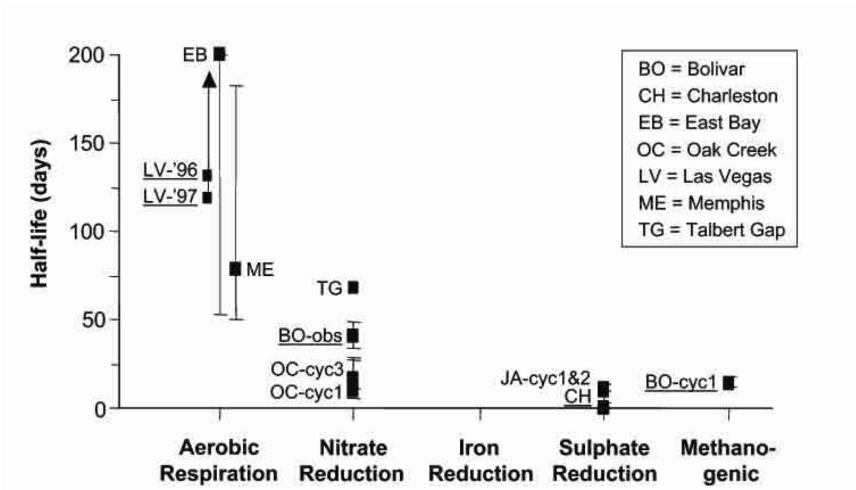


FIGURE 4-6 The observed transformation rate of THMs estimated during aquifer storage is related to the redox conditions of the system. Rates are very slow in aerobic systems (long half-life), intermediate in nitrate reducing systems, and most rapid in anaerobic (sulfate reducing or methanogenic) systems. The authors point out that although the Oak Creek site results seem to indicate an exception to the trend, the redox state determined at this site was of "low reliability." Figure from Pavelic et al., 2006. Reprinted, with permission, from Pavelic et al. (2006). Copyright 2006 by Elsevier Limited.

under identical geochemical conditions, brominated THMs are less persistent than their chlorinated counterparts (Pavelic et al., 2005a, b, 2006).

Although limited, the available information suggests that HAAs are not persistent during aquifer storage. Monitoring of the aerobic system at the Las Vegas field site showed that although the total HAA concentration increased in samples collected soon after recharge, water recovered after relatively short storage periods of as little as 50 days contained no detectable HAAs (Thomas et al., 2000). Similar results (e.g., detectable HAAs in recharge water or recovered water) were observed at the Bolivar field site, which has very different in situ geochemical conditions (Pavelic et al., 2005c). Also consistent between these two field sites is the observation that the total HAA concentration declined more rapidly than did the total THM concentration. Limited controlled laboratory studies support the interpretation that biotransformation is the primary mechanism by which HAA concentrations are attenuated during aquifer storage. Studies using either tri- or monochloroacetic acid showed that HAAs can be used as both a carbon and an energy source by cultured microorganisms (McRae et al., 2004; Torz and Beschkov, 2005). Monochloroacetic acid was mineralized (transformed to CO_2) in aquifer microcosms under both aerobic and anaerobic conditions, while it was persistent in abiotic control microcosms (Landmeyer et al., 2000). Trichloroacetic acid has also been shown to be persistent in abiotic iron reducing systems (in the presence of iron oxide minerals and Fe(II)) (Chun et al., 2005). Trichloroacetic acid transformation can also produce chloroform (Xiang et al., 2005).

Overall, THM and HAA formation and attenuation are observed in aquifer storage systems studied to date in which residual disinfectant is also present. Water quality improvement with respect to these compounds is observed in many cases. The rate and extent of attenuation of DBPs depends on the geochemical conditions within the aquifer and on the chemical properties of the compound of concern. Among THMs and HAAs, chloroform is generally the most persistent. Persistence varies not only between aquifer systems, but also in the different redox conditions that are present within a particular storage system and can change in both space and time. For example, in an aquifer storage system with degradable DOC and solids containing available iron oxide minerals (such that iron reducing conditions are dominant over methanogenic conditions), chloroform persistence is expected while more brominated THMs are likely to be transformed. Outstanding issues for DBP behavior in MUS include the following:

- Improving predictive capability for DBP degradation rate associated with various geochemical conditions during MUS;
- Predicting variable geochemical conditions that will occur in space and time within a particular MUS system; and
- Providing a more thorough assessment of the distribution of DBP transformation products resulting from storage under different geochemical conditions.

Pharmaceuticals, Personal Care Products and Other Emerging (Presently Unregulated) Compounds

The occurrence and significance of anthropogenic compounds in surface waters impacted by reclaimed water discharges in the United States is described by Kolpin et al. (2002). These workers (Kolpin et al., 2002) sampled 139 streams in the United States and analyzed the samples for 93 organic waste contaminants and a wide range of PPCPs. They identified widespread occurrence of many of these compounds at trace levels that resulted in increased concerns about the safety of surface drinking water supplies. The widespread occurrence of these compounds in the United States and Europe was previously discussed by Daughton and Ternes (1999) and Ternes and Joss (2006); their studies suggest that while impacts to aquatic life and other environmental impacts are possible, the concentrations of pharmaceuticals observed are too low to have a defined impact on human health. Nevertheless, concerns about these emerging contaminants have resulted in active research on the fate and transport of these compounds in the environment, including the subsurface environment.

Note that the analytical methods for PPCPs often involve the use of high-performance liquid chromatography coupled with mass spectrometry (HPLC-MS) and that a limited number of laboratories equipped to analyze environmental samples for concentrations in the nanogram-per-liter range. Furthermore, the laboratories must have specific licenses to handle regulated pharmaceuticals.

Research on the fate of PPCPs during subsurface transport in Europe has focused on bank filtration (Heberer et al., 2001). Several monitoring studies carried out in Berlin, Germany, between 1996 and 2000 identified pharmaceuticals such as clofibric acid, diclofenac, ibuprofen, propyphenazone, primidone, and carbamazepine at individual concentrations up to the microgram-per-liter level in influent and effluent samples from wastewater treatment plants (WWTPs) and in all surface water samples collected downstream from the WWTPs (Heberer, 2002). Under recharge conditions, several compounds including primidone and carbamazepine were also found at individual concentrations up to 7.3 μg/L in samples collected from the underlying groundwater. A few of the compounds were also identified at the nanogram-per-liter level in tap water samples from Berlin, where bank filtration is used to purify surface water supplies.

Common interests in the subsurface persistence and mobility of PPCPs in source waters impacted by treated wastewater led to a cooperative study between European and American Researchers. The research focused on the use of reclaimed water as source water to recharge basins (United States) and the use of sewage-contaminated surface waters in bank filtration systems (Europe). The principal attenuating processes were biological transformation and sorption. The occurrence of these processes differed depending on compound structure, soil, and biogeochemical conditions.

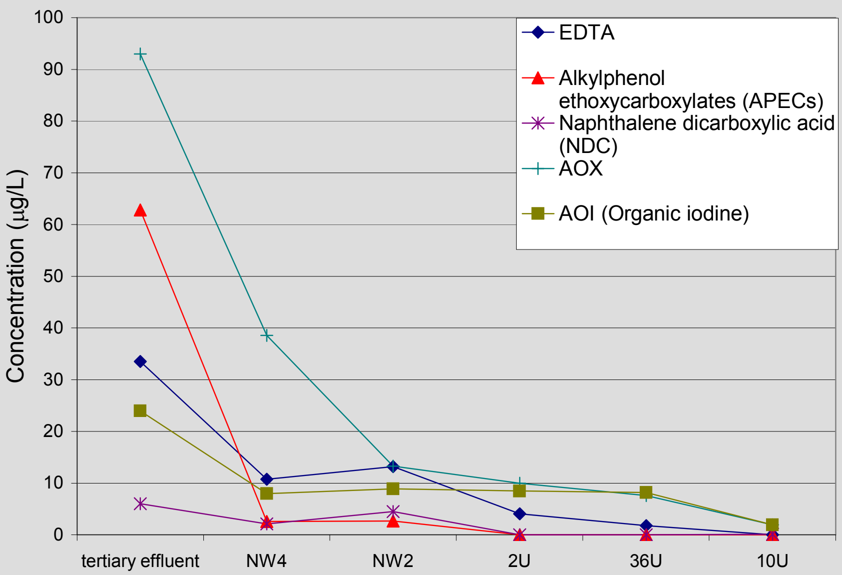
Four different classes of pharmaceutical compounds were selected for this study based on analytical and sample volume limitations. Details of the analytical methods and sampling methodology were presented in Drewes and Shore (2002). The study identified that recharge basins in the southwestern United States and bank filtration systems in Europe attenuated synthetic organic compounds with almost identical results. The major difference was that the concentrations were approximately three times higher in Europe, which can be explained by Europe's more efficient water use, which results in less dilution. Illustrative results from the U.S. study are provided in Box 4-5 and Figure 4-7. The majority of compounds measured were attenuated. Attempts to develop a time-distance relationship for the attenuation processes were successful for specific types of systems such as flow through porous media in sand and gravel aquifers. As a result of this research, certain anthropogenic compounds were determined to be persistent in most underground storage systems; however, the health effects associated with these compounds at nanogram-per-liter concentrations were not assessed. The characteristics common to those compounds that

BOX 4-5
Fate of Selected Trace Organic Compounds During Long-Term Storage at the Mesa Northwest Water Reclamation Plant (NWWRP)

Analysis of water samples reflecting long-term subsurface storage following recharge using surface spreading illustrates removal of most trace organic compounds and persistence of a few compounds. The behavior of selected trace organics during underground storage was studied to identify and quantify processes that affect organic contaminant attenuation during subsurface transport.

Initial research activities focused on the fate of the following compounds at microgram-per-liter concentrations in source water: clofibric acid, surfactants such as alkylphenolethoxylates, DBPs, nitrilotriacetic acid (NTA), and ethylenediaminetetraacetic acid (EDTA) (Montgomery-Brown et al., 2003). As analytical techniques improved to detect compounds at nanogram-per-liter concentrations, concern about PPCPs and endocrine disrupting compounds (EDCs) led to additional research on these emerging contaminants of concern. Figure 4-7(a) shows that alkylphenol ethoxycarboxylates (APECs) and EDTA were removed to detection limits after approximately one year of travel time. The fates of adsorbable organic halides (AOX) and adsorbable organic iodine (AOI) are also presented in Figure 4-7a (Fox et al., 2001). After long-term treatment, AOX concentrations were at the same level as AOI concentrations, which implies that the adsorbable chlorinated and brominated compounds were removed to background concentrations and that the persistent AOX were iodated. This work on total organic halides suggests that the chlorinated DBPs were efficiently attenuated during subsurface transport at this field site.

Consistent with other subsurface transport studies, carbamazepine and primidone were persistent at the Mesa site, as shown in Figure 4-7(b). The combined results of this study illustrate how some PPCPs can persist under conditions that are ideal for biotransformation of many trace organic chemicals.



(a)

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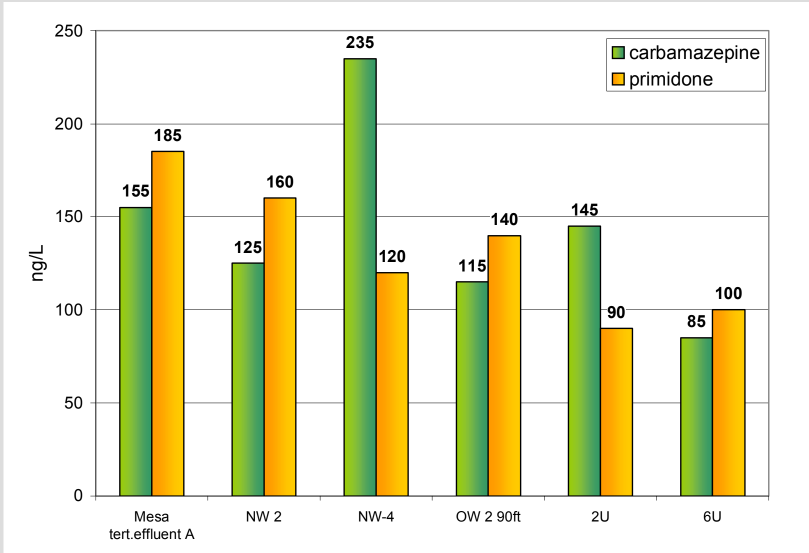
BOX 4-5 Continued



(b)

FIGURE 4-7 The fate of several trace organic compounds during long-term soil-aquifer treatment at the Mesa Northwest Water Reclamation Plant. (a) The adsorbable organic iodine (AOI) persists, while several other compounds (NDC, APECs) are removed. (b) Carbamazepine and primidone persist. NOTE: HPEC=alkylphenol ethoxycarboxylates; NDC= naphthalene dicarboxylic acid

are not attenuated are that they are hydrophilic (polar) and they have structural features that prevent enzymatic attack and render them resistant to biodegradation. Examples of persistent compounds are antidepressant drugs such as carbamazepine and primodone; the fire retardant tri(2-chloroethyl) phosphate; the mosquito repellant DEET (*N*, *N*-diethyl-*m*-toluamide) and organic iodine, the residual of an X-ray contrast agent (Heberer, 2002; Clara et al., 2004). The persistence of carbamazepine has led researchers to suggest using it as a universal indicator of anthropogenic contamination (Clara et al., 2004).

Another field example showing attenuation of PPCPs during infiltration and storage is from the Tucson Underground Storage and Recovery Facility. The fate of five analgesics was examined. These analgesics are removed at many wastewater treatment plants, but they were not removed prior to groundwater recharge at the Tucson facility. Figure 4-8 shows that these compounds were present in the effluent source water but were reduced to nondetectable concentrations in water sampled directly below the recharge basin at a point representing a travel time of less than one month. In this recharge system, PPCPs were attenuated during recharge and storage.

Concern about endocrine disrupting compounds (EDCs) has led to research on the fate of estrogenic hormones and alkyphenols known to exhibit estrogenic activity. As expected by Heberer (2002), these compounds are efficiently attenuated during subsurface transport. These compounds have been demonstrated to accumulate in the upper soil layers of recharge basins; however, the adsorbed compounds are biodegraded and their accumulation levels appear to reach a steady concentration with no risk of breakthrough.

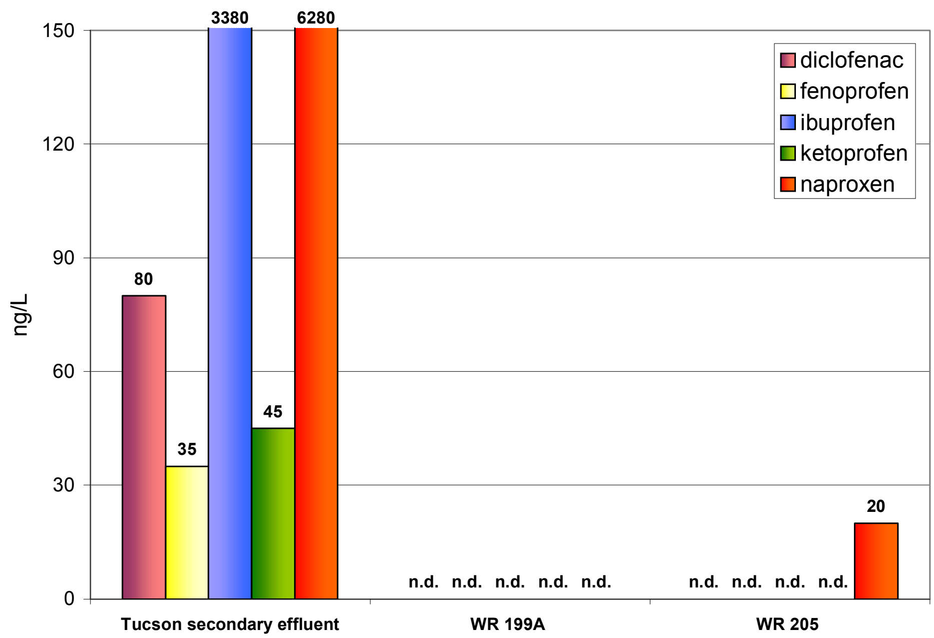


FIGURE 4-8 The fate of analgesics during groundwater recharge at the Tucson Underground Storage and Recovery Facility. Sampling point WR199A was located directly below the basin with a travel time of less than one month and WR206 was located downgradient of the recharge basin.

There is an emerging concern over the disinfection by-product, *N*-nitrosodimethylamine. It is a particular concern for projects with reclaimed water supplies. Reclaimed waters contain SMPs with elevated levels of organic nitrogen that may result in greater production of nitrogenous DBPs, such as NDMA, compared to DBPs formed from NOM. Research on recharge basins has demonstrated that NDMA is effectively attenuated by several mechanisms during groundwater recharge. Because NDMA is light- sensitive, sunlight may reduce its concentrations in recharge basins prior to subsurface transport. Both aerobic and anaerobic microbial mineralization of NDMA has been observed in soils obtained from recharge basins, and these mechanisms may be a substantial component of NDMA attenuation in soils underlying groundwater recharge facilities. The presence of NDMA in the product water from indirect potable reuse systems using recharge basins has not been observed, although concentrations in excess of 1,000 ng/L have been applied at some sites.

The shutdown of two municipal water supply wells in Orange County, California, in response to aquifer NDMA contamination amply illustrated the risks associated with direct aquifer recharge and the need to evaluate the natural attenuation capacity of the soil environments that are involved in groundwater recharge and storage operations. While factors including dilution, dispersion, and adsorption are expected to contribute to NDMA attenuation during wastewater reclamation, biodegradation is anticipated to be the primary mechanism of contaminant destruction in surface and vadose zone soils. The breakthrough of NDMA in recharge well systems may occur if the attenuation mechanisms in the aquifer are not effective. When recharge wells are used, there is no exposure to sunlight, eliminating an abiotic destructive mechanism. The Orange County Water District Water Factory 21 uses reverse osmosis for treatment prior to injection. Reverse osmosis removes almost all organic carbon with the exception of low-molecular-weight nonpolar compounds such as NDMA and 1,4-dioxane. By removing almost all nutrients and organic carbon prior to recharge, the biological attenuation mechanisms in an aquifer may be limited. Consequently, compounds present at very low (nanogram-per-liter) concentrations incapable of supporting microbial metabolism are unlikely to be removed during subsurface transport.

Metals and Metalloids

The speciation of metals and metalloids (within the text of this section, the term ‘metals’ is used to mean both metals and metalloids) affects their mobility and toxicity. Unlike organic contaminants that can be mineralized to innocuous products, metals cannot be eliminated from an MUS system although they can be rendered relatively immobile by either strong sorption or precipitation reactions that transfer the metal from the mobile dissolved phase to the solid phase. Also, in contrast to most organics, metal contamination of stored water can occur in situ by changes in geochemical conditions. For example, the release of arsenic, cobalt, iron, manganese, molybdenum, nickel, vanadium, and uranium from the aquifer solids to the recharged water has been documented at several aquifer storage and recovery facilities in Florida (Arthur et al., 2005).

Metals that form cationic dissolved species, including cadmium, copper, lead, and zinc, are mobile in acidic environments. These metals form relatively insoluble carbonate, hydroxide, or sulfide minerals at moderate to high pH. Sorption onto mineral surfaces at circumneutral pH also reduces the mobility of these metals. Because common hydroxide and silicate mineral surfaces carry a negative charge at near-neutral pH conditions, they will strongly sorb many cationic metals. In contrast, in acidic systems, cationic metal ions tend not to sorb and tend to be very mobile.

Metals that form anions or oxyanions in solution are often relatively mobile. The sorption behavior of each metal oxyanion is dependent on system conditions (e.g., pH, Eh, competing constituent concentrations). Metals that can take on multiple redox states (e.g., iron, arsenic) typically form relatively insoluble mineral precipitates or coprecipitate with iron and sulfide under reducing conditions.

Arsenic presents a particularly complex example that is relevant to MUS (Boxes 4-6 and 4-7). Arsenic contamination of stored water by release from the aquifer has been associated with artificial recharge in Florida (Box 4-6 and Figure 4-9), Wisconsin and the Netherlands (e.g., Arthur et al., 2001, 2005; Johnson et al., 2004; Roth, 2004; Stuyfzand, 1998). Arsenic exists naturally in the −3, 0, +1, +3, and +5 oxidation states. Arsenic speciation and dissolved concentrations depend on geochemical conditions, including redox conditions, pH, organic matter content, the presence of iron oxides, ions that compete for adsorption sites, solution composition, aquifer mineralogy, and reaction kinetics (Nordstrom, 2002; Smedley and Kinniburgh, 2002; Welch et al., 2000).

Dissolved groundwater arsenic is usually present in the inorganic forms arsenite As(III) and arsenate As(V) (Welch, 2000), which differ in mobility and toxicity. Arsenite (As^{3+}) is more toxic than arsenate (As^{5+}). Arsenate (As(V); $H_nAsO_4^{n-3}$), which dominates in aerobic environments, generally adsorbs strongly to iron oxides, clays, or silicates in soils and sediments (i.e., Hounslow, 1980; Lin and Puls, 2003; Meng et al., 2002). Arsenite (As(III), $H_nAsO_3^{n-3}$) is the dominant arsenic species in anaerobic waters. Arsenite can also sorb strongly to iron (hydr)oxides and iron sulfide minerals, but it has a narrow adsorption envelope centered around pH 7 and does not partition extensively onto aluminum hydroxide or aluminosilicate minerals (e.g., kaolinite). Arsenic is mobilized under iron reducing conditions, because iron oxides dissolve and release adsorbed arsenic (Smedley and Kinniburgh, 2002). Thus, in nonsulfidic systems where ferric (hydr)oxides are absent or undergoing degradation or where the pH deviates appreciably from neutrality, one can expect arsenic to partition to the solution phase. However if sulfate reduction occurs, the H_2S produced can result in the formation of arsenic-bearing minerals including sulfides. In these circumstances, the mobility of arsenic may be reduced by precipitation reactions.

BOX 4-6
Arsenic Release from Aquifer Solids to ASR Recharged Water in the Floridan Aquifer System, Florida

In Florida, 17 ASR facilities that recharge source water into an underground source of drinking water (USDW; see Box 4-1 and Chapter 5) have sufficient water quality data to assess arsenic behavior. Arsenic leaches from the limestone aquifer in 13 of these ASR systems at concentrations exceeding the drinking water standard. The release of arsenic and other metals occurs in response to geochemical differences between the source (recharged) water and in situ native groundwater and interaction with the aquifer matrix. Field testing has elucidated the dynamics of arsenic release from the aquifer, and complementary laboratory work has identified the solid phases containing arsenic (e.g., pyrite; see Box 4-7) and the leachability of arsenic from these phases under varying redox conditions.

Contrasts in the patterns of arsenic and calcium concentrations during the recharge and recovery phases of the field tests using a single well confirm that the elevated arsenic in the recovered water was released from the aquifer solids (Figure 4-9). Arsenic in both the native groundwater and the recharged water is below the water quality standard of 10 μg/L. The calcium concentrations in the recharged and native waters differ substantially. The increase in calcium concentration indicates the transition (mixing) between injected and native groundwater in the recovery phase of the field tests shown in Figure 4-8. The peak arsenic concentration observed in the second test cycle is substantially lower than that observed in the first test, demonstrating a decline in arsenic mobilization associated with repeated recharge, storage, and recovery events of comparable size and duration. The observation of damped chemical release behavior with repeated recharge, storage, and recovery tests is known as ”conditioning” and is described further later in the chapter.

Ongoing research in Florida suggests that arsenic may not migrate far from the ASR well; however, results are site-specific and depend on the local hydrogeologic and hydrogeochemical setting. For example, of the 13 ASR systems referenced above, arsenic has been detected above 10 μg/L in monitor wells at 3 of these facilities.

The Florida regulatory community, recognizing that reasonable assurance of protection of the USDW may be accomplished with appropriate operational practices and monitoring, has proposed clarifying language to the EPA on the subject of arsenic and ASR in Florida. At present, an EPA workgroup is addressing the issue. In the meantime, several

Nitrate may also affect redox geochemistry in groundwater pertinent to arsenic speciation. Similar to the behavior in the presence of oxygen, nitrate mixed with waters containing reduced arsenic and iron leads to oxidation of As(III) to As(V). Under such conditions, nitrate is also expected to oxidize Fe(II) to Fe(III), reducing arsenic mobility.

The TOC, including both dissolved and particulate forms, can also affect the mobility and speciation of some metalloids, such as arsenic and mercury, through the formation of organic complexes or organic species. Organic mercury species are particularly important because they are more toxic than inorganic forms to both humans and aquatic organisms. Organic mercury is more mobile than the inorganic forms in soils and is known to bioaccumulate in ecosystems. Unlike arsenic and some of the other metals discussed in detail above, mercury contamination is generally derived from anthropogenic activities. It is most likely to be added to an MUS system through recharge of surface waters because of its pervasive distribution in the surficial environment at low concentrations, usually in the inorganic and less toxic form. Methylation of Mercury municipalities that were planning to implement ASR await the regulatory outcome. In southwestern Florida, some facilities are considering water resource alternatives, such as treatment of brackish water with reverse osmosis in order to meet projected demands.

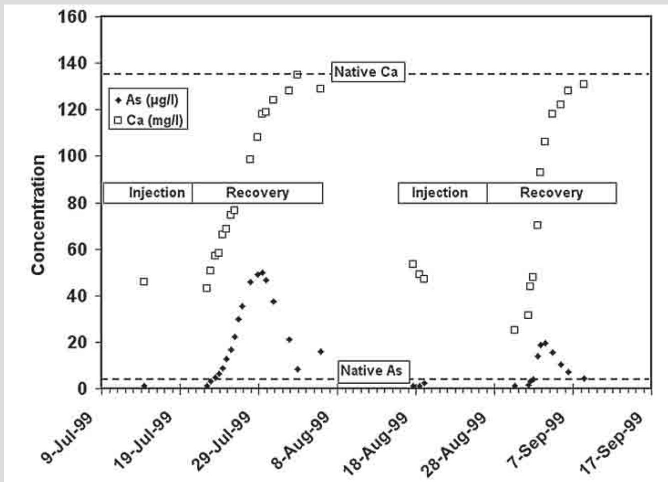


FIGURE 4-9 Field injection and recovery tests in a limestone aquifer in Florida showing that arsenic is released to injected water during storage. Calcium serves as a tracer of injected, mixed, and native groundwaters in this example.

(conversion from inorganic to organic forms) is known to occur under sulfate reducing geochemical conditions. Therefore, mercury methylation would be favored in a sulfate reducing MUS system that uses injection of water containing trace concentrations of mercury. Preliminary results suggest that such processes may be problematic during storage in Florida MUS systems (Hodo et al., 2004).

Case Studies with Microorganisms

Recent studies have been undertaken on the resistant protozoan *Cryptosporidium* in native surface waters and groundwaters that were being used for aquifer storage and recovery in Florida. Bench-scale survival studies with *Cryptosporidium parvum* were conducted in representative aquifer and reservoir waters of Florida. *C. parvum* inactivation rates ranged from 0.0088 $\log_{10}d^{-1}$ at 5° C to −0.20 $\log_{10}d^{-1}$at 30°C. Temperature, water type, and the interaction of these factors had statistically significant effects on *C. parvum* survival.

The Viable Nonculturable Issue: The Need for Application of Advanced Molecular Techniques

The measurement and cultivation of many bacteria collected from environmental samples may become difficult due to their entrance into the viable but nonculturable (VBNC) state. It has been known for some time that bacteria may retain viability and can begin to replicate under appropriate conditions but remain unculturable on routine bacteriological media (Oliver, 2002). This status is influenced by a number of environmental conditions that may stress the organism, including temperature, changes in nutrient availability (Oliver, 2002), and other factors such as increased oxygen tension and exposure to antibiotics. One of the emerging microbial contaminants (on the EPA Contaminant Candidate List) that has a particular association with groundwater is the bacterium *Helicobacter pylori,* a known cause of ulcers. Nilsson et al. (2002) reported that *H. pylori* changed their morphology when exposed to water for prolonged periods of time, transforming to a coccoid form and entering into a viable but nonculturable state. The coccoid form may be responsible for waterborne transmission (Hulten et al., 1998).

This must be recognized as an issue for MUS. Noncultivatable techniques should be used in the future for assessment of water quality and risk. Figure 4-10 shows the decrease in *Helicobacter pylori* concentrations in groundwater over time at two temperatures by routine cultivation techniques and by a new genetic method (polymerase chain reaction [PCR]; Nayak and Rose, 2007). Many microbial contaminants will not be detectable unless these advanced methods are used.

In addition a number of enteric viruses such as norovirus, are not cultivatable. Thus risks to waters, groundwater, and MUS systems cannot be assessed adequately without application of the new methods. PCR is the most popular molecular technique to date. Any new pathogen can be detected now with PCR once part of its genetic code has been identified. PCR is an enzyme-driven method for amplifying short regions of DNA in vitro. PCR detects live and dead particles, can detect microorganisms that we do not know how to cultivate, is highly specific, and obtains results generally in 24 to 48 hours. Molecular tools for environmental microbial assays are still under development but have promising capabilities for the next generation. Although current water quality standards and guidelines are based on indicator microbes along with a few pathogens for drinking water, there is little effort to apply these new techniques for better assessment of surface, ground, and MUS waters. In order to move beyond the use of conventional methods, it will be necessary for scientists in academia, industry, and government agencies to collaborate and to mobilize efforts to improve the application of these tools within risk and regulatory frameworks.

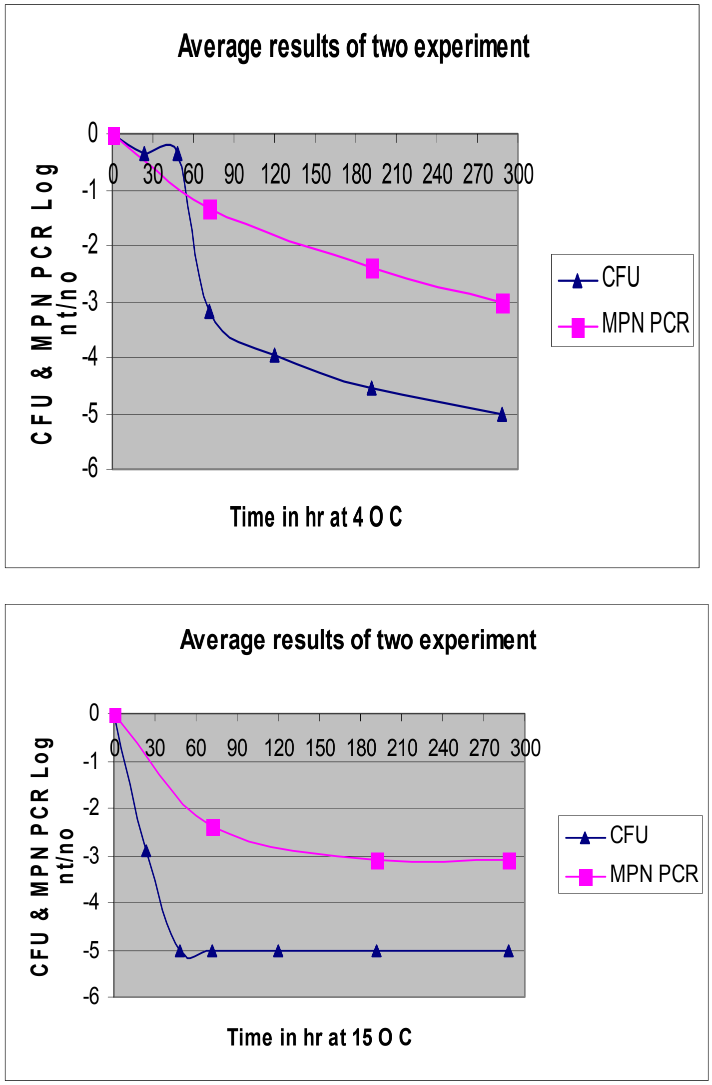


FIGURE 4-10 Use of molecular techniques compared to cultivation for characterization of *Helicobacter* in groundwaters. Reprinted, with permission, from Nayak and Rose (2007). Copyright 2007 by Blackwell Publishing Ltd.

EFFECTS OF WATER QUALITY ON MUS PERFORMANCE

Aquifer Clogging and Dissolution

Clogging, a reduction in permeability caused by physical filling of pore space in the aquifer media, is an important issue with regard to MUS system performance. Although generally interrelated, the processes that contribute to clogging can be divided into three categories: physical, chemical, and biological. Depending on the MUS system, the most dominant process may differ. For example, in a recharge basin, physical and biological clogging may predominate. In a recharge or ASR well, however, chemical and biological clogging may be the greatest cause for concern. For example, encrusting precipitates or biofilm growth on well screens will reduce system performance. Clogging depends on the interactions among aquifer, source, and receiving water properties and on operational variables, including aquifer matrix properties (e.g., effective porosity, lithology, mineralogy, cation exchange capacity), source and native groundwater quality (e.g., redox conditions; dissolved metals, carbon, and other nutrients), microbial activity (e.g., microbial-induced mineral precipitation, biomass production), pumping or infiltration rates, temperature, and light intensity (in recharge basins).

Physical clogging involves reduction of permeability through buildup of particulate matter or gas entrainment. Sediment ("cake" or "sludge") buildup can occur by filtration or straining of suspended solids in water and is hence dependent on the concentration and composition of the suspended solids, recharge or infiltration rates, and durations. For example, Konikow et al. (2001) showed that physical clogging of the aquifer formation can result from mobilization of clay particles when a brine aquifer is recharged with fresh water. A more common example of physical clogging is presented by Pavelic et al. (2007) who describe clogging resulting from suspended matter in the recharge water of an ASR system. Clay swelling is yet another contributing factor to physical clogging.

Gas entrainment, although physical in the context of reducing permeability by decreasing connected water-filled pore space, is caused by either biotic or abiotic chemical reactions. It is noteworthy that the gas is often not air. Bouwer and Rice (1989), for example, report microbially induced denitrification gases as a causative factor in clogging. Temperature and pressure differences may also lead to gas exsolution and clogging. Although pressure is less likely to be an issue with a recharge well, mixing of waters (one being oxygen-rich) of contrasting temperatures may yield degassing, in which small bubbles may form within the matrix porosity. In addition, gas entrainment can occur due to cascading water in a recharge well.

Chemical clogging involves hydrogeochemical reactions that result in mineral or colloid (i.e., gelatinous) precipitation. Some of the more common precipitates include calcite, gypsum, phosphates, and iron and manganese oxides or hydroxides. Moorman et al. (2002), for example, report the following chemical clogging constituents during recharge of pre-treated River Rhine water: iron hydroxides, ferric hydroxiphosphates, and secondary deposits of hydroxyapatites. They also observed a biological clogging factor—filamentous iron oxidizing bacteria. Factors involved in chemical clogging include source and native groundwater composition, chemical effects of mixing of these waters, and water-rock interactions. Redox conditions, acid-base reactions, and biogeochemical reactions are also important.

It is noteworthy that geochemical dissolution reactions may also offset the effects of clogging by increasing matrix permeability and porosity. Increased porosity and permeability have been documented during field scale recharge experiments in calcareous (carbonate) aquifers in Australia (Herczeg et al., 2004; Pavelic et al., 2007) and South Carolina (Mirecki, 2004). For example, a mixture of two waters in equilibrium with respect to calcite but with different carbon dioxide concentrations (reflected in part by different pH or acidity) can form a new solution that is undersaturated with respect to calcite and therefore chemically aggressive to carbonate rocks. This "mixing corrosion" phenomenon also occurs along certain freshwater-saline water interfaces. Calcite mineral dissolution can occur as an acid neutralization reaction in response to elevated acid concentrations created by the source water through either degradation of organic matter in the source water that increases the carbonic acid concentration or oxidation of aquifer sulfide minerals by dissolved oxygen in aerobic source water that produces sulfuric acid. In any of these cases, permeability of the aquifer matrix may increase over time.

Microbial growth and accumulation of extracellular polymers lead to biological clogging, which is also referred to as "biofouling," "bacterial clogging," or "bioclogging." Other forms of biomass in source waters that contribute to clogging include algae and diatoms. MUS systems involving the recharge of relatively nutrient-rich waters (e.g., reclaimed or wetland-treated water) containing nitrates, phosphates, and/or dissolved or particulate organic carbon will stimulate microbially mediated redox reactions and biomass growth. Although bioclogging is a well known phenomenon in water filtration, a review of the topic by Baveye and others (1998) identifies critical needs for more mechanistic understanding and the capacity for predictive modeling.

Microbially mediated redox reactions can have an effect on redox conditions in the MUS storage zone, which may consequently affect aquifer permeability. For example, during ASR recovery where native groundwater may displace recharge or transition water, sulfate reduction creates reducing conditions in the aquifer and produces dissolved H_2S. In such conditions, and with sufficient time, dissolved Fe(II) is favored to precipitate as pyrite. Formation of solid products can contribute to clogging.

Conditioning Processes

Aquifer or storage zone conditioning broadly refers to gradual improvements in performance or water quality characteristics of an MUS system after successive recharge periods or cycle tests. Recovery efficiency (see Chapter 3) is an example of an MUS performance measure that may exhibit improvement upon repeated ASR cycle testing. Reese (2002) noted recovery efficiency improvements for some, but not all, ASR wells in southern Florida as the number of completed cycle tests increased. In MUS systems, however, conditioning is more widely considered to be associated with water quality improvements. A comprehensive report on water quality improvements during ASR (Dillon and Toze, 2005) demonstrated the ability of certain aquifers to attenuate DBPs, EDCs, and pathogens, depending on various physical, chemical, and biological parameters/processes (e.g., redox conditions, microbial activity). As noted in Box 4-6, there is an indication of conditioning with respect to metal mobilization during ASR. In recognition of the conditioning process, the South Australia Environment Protection Authority (2004) Code of Practice for Aquifer Storage and Recovery allows for designation of an attenuation zone (see also "Travel Time or Residence Time Criteria" Chapter 5). The code strongly recommends that a monitoring program be designed and interpreted by a suitably qualified professional hydrogeologist to demonstrate that contaminants are reduced by physiochemical and microbiological processes in the designated attenuation zone.

Although a particular aquifer may exhibit the ability to condition or attenuate a chemical constituent, a complete understanding of the processes that contribute to the conditioning effect is important with regard to understanding the conditioning capacity. For example, if an aquifer has a capacity to sorb a particular constituent of concern, the system may reach a threshold above which sorption can no longer occur; therefore, the constituent may become a renewed water quality issue. Moreover, in some MUS systems, especially those with preferential flowpaths, chemically reactive waters may migrate beyond a delineated zone. In such cases, the placement of monitor wells, sampling frequency, and parameter selection becomes even more important.

Conditioning processes must also be considered in context of time and scale. In Box 4-6, for example, attenuation of arsenic is indicated based on data collected during cycle testing. These cycle test volumes may not reflect those anticipated during full-scale operation of the system. As a result, scaling up recharge volumes may yield additional arsenic concentrations due to exposure to previously unaffected aquifer media, and therefore the baseline of conditioning would be reset. Natural attenuation would not likely occur until completion of repeated full-scale cycle testing.

TOOLS TO PREDICT WATER QUALITY AND AQUIFER CHANGES DURING MUS

Multiple approaches can be taken to assess potential or existing groundwater contamination resulting from MUS activities. These approaches begin with characterization of: (1) waters (source, mixed, and native groundwater; chemical and physical parameters); (2) aquifer media (rocks and sediments; lithogeochemistry, texture, hydrogeologic properties, mineralogy and mineral chemistry [including trace constituents]); and (3) microbial populations (source, native subsurface; fate and transport) within the MUS system. Laboratory experiments allow assessment of chemical and/or biological reactions under controlled conditions. In experiments such as bench-scale and column studies, it is possible to examine the effects of geochemical variables on water-rock reactions and to characterize reaction rates, pathways, and changes in water quality. Through identification of these processes and pathways, improvements in design and implementation of MUS systems may be realized.

An obvious limitation of laboratory experimental systems is that they provide results limited in applicability to the specific study conditions. Hence, detailed understanding of the field system is required to determine the applicability and/or design of complementary laboratory experiments. Field testing, including detailed analysis of operating field systems, provides essential information on contaminant fate under complex operational conditions. It is difficult, however, to determine basic reaction pathways and/or controlling conditions based on field experiments alone.

Geochemical modeling facilitates overall MUS system characterization and enhances the ability to develop predictive tools. These models, which estimate processes and their effects in the natural system, are more accurate when based on or validated with site-specific field data. As a result, a combination of laboratory and field-scale assessments,[2] coupled with geochemical modeling, yields a more robust and applicable characterization of the overall MUS system in the context of biological, hydrogeological, and hydrochemical processes.

Batch and Column Scale Studies

Laboratory experiments can identify potential changes in water quality and permeability at the field scale. Not only can the changes be approximated, but in optimal conditions, the results of these studies may be calibrated with field data or geochemical models to become predictive tools. A wide variety of

[2] A perception issue exists among many regulatory and municipal agencies with regard to "scientific experiments" and "research." At issue is the connotation that research should be completed primarily in an academic environment, despite the fact that this particular type of research is applied and will be used in science-based policy and decision making. The term "assessment," which can mean the same as applied research, has been found to be more acceptable among the regulatory community.

methodologies for these laboratory-scale column and batch studies exists. In general terms, a column study is an experimental system designed to allow water to flow through aquifer media, during which time changes in water quality, injection pressure, or permeability can be monitored. The simplest experimental design includes use of representative recharge water with all physical and chemical variables (i.e., TDS, pH, temperature) held constant. Multiple columns may be used to assess the effects of heterogeneity in the aquifer media or the source water. For example, the source water may have been collected from different localities or during different seasons to reflect spatial or temporal variability. On the other hand, the water may represent a single source, with artificial adjustments made to pH or ORP. Aquifer media samples are generally preferred in the form of a core. Column studies are designed such that flow through the rock-sediment column is intergranular. Changes in column flow rates, reflecting permeability changes, also indicate changes in water chemistry as chemical constituents in the source water are either gained (sorption-precipitation) or lost (desorption-dissolution). Intentional changes in the flow-through water chemistry (i.e., pH adjustments) facilitate assessment of physical, chemical, and biological clogging (see "Aquifer Clogging and Dissolution"). Results of column studies can be applied toward optimization of full-scale MUS operations.

Batch or bench-scale studies are another means of assessing water quality changes due to water-rock-microbial interactions. These experimental designs are not often "flow-through" systems, but rather static "closed-system" conditions allowing "snapshot" assessment of leachability (i.e., one sample after 18 hours) or analyzing a series of samples to assess water quality changes through time. Bench-scale leaching methods may include the EPA synthetic precipitation leaching procedure (SPLP; EPA Method 1312) and the U.S. Geological Survey (USGS) Field Leach Test (Hageman and Briggs, 2000). More complex leaching studies include application of different waters as "leaching agents," again to assess spatial or temporal variability or to simulate MUS operational conditions (e.g., use of native and source water). During ASR, for example, the native aquifer storage zone is often a hydrochemically reducing environment: low dissolved oxygen concentrations and negative oxidation-reduction potential. Source waters often reflect oxidizing conditions and near-DO saturation. During recharge of these waters into the native aquifer, disequilibrium occurs when minerals once stable in the reduced environment become unstable, releasing metals into the recharged water through desorption or dissolution. Bench-scale studies can be designed to provide an estimate of these water-rock reactions to changing redox and DO conditions.

Box 4-7 and Figures 4-11 through 4-13 illustrate of how a variety of laboratory tests can be used to evaluate the reaction processes controlling reactions during recharge, storage, or recovery.

Not only are column and bench studies useful for characterizing hydrogeochemical or even microbiological processes that may affect MUS water quality, these laboratory studies may also be employed to test hypotheses that may mitigate unfavorable water quality changes. In the pyrite oxidation example, DO

BOX 4-7
Laboratory Experiments and Aquifer Geochemical Characterization to Understand the Source of Arsenic to Injected Water in Florida ASR

Laboratory experiments and geochemical characterization have been used to better understand the processes causing the patterns of arsenic release from aquifer solids during field experiments (pilot ASR cycles) as illustrated in Box 4-5. A goal is to reduce contaminant release through improved design and/or operations based on knowledge of the arsenic forms and leachability. The laboratory work summarized here (from Arthur et al., 2007) identified the solid phases containing arsenic, the leachability of arsenic from these phases, and the potential dynamics of arsenic release or storage (sorption) associated with varying redox conditions.

Batch reactors containing aquifer solids were used to examine the geochemical conditions that control the release of arsenic and other trace metals to injected water. The aquifer solids, carbonate rocks from the Floridan Aquifer System, were crushed core segments that were trimmed of their exposed drilling surfaces. Figure 4-11 summarizes results of the bench-scale leaching study that exposed the aquifer solids to native groundwater (phase 1), followed by source water (phase 2) intended for use in an ASR system. Nitrogen gas was used in the sealed reactors' head space to achieve low-DO (<0.4 mg/L) conditions during phase 1 and 2a of the experiment; whereas DO was saturated (>7 mg/L) during phase 2b. Despite the relatively low-DO conditions, arsenic mobilization is observed and is attributed to pyrite oxidation (equation) owing to sufficient DO and relatively high ORP:

$$FeS_2 + 7/2\ O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+.$$

Leaching or dissolution of pyrite oxidation products that formed during core storage may have also influenced initial mobilization in the batch reactors. Arsenic was released with declining Eh (ORP) under low dissolved oxygen (LDO) conditions with either native groundwater (NGW) or treated surface water (SW). However, the arsenic concentration declined rapidly following the introduction of high DO (HDO) conditions in the batch systems (phase 2b in Figure 4-11). The authors posit that during phase 2b, the dissolved arsenic sorbed to hydrous ferrous oxide (HFO) precipitate. The removal of available iron during the high DO phase of the experiments, as well as follow-up testing, supports this mechanism.

During storage and recovery in an ASR system, native reducing conditions are often drawn toward the ASR well. In the event that HFOs have indeed sorbed arsenic as suggested in the bench-scale study, reducing conditions during ASR recovery may again release the arsenic into solution through reductive desorption or dissolution of the HFOs (Pieter Stuyfzand, personal communication, 2006; Vanderzalm et al., 2007), similar to a mechanism reported by Gotkowitz et al. (2004) for pumping wells in a confined aquifer.

As determined by sequential extraction, leachable arsenic is associated primarily with sulfide minerals and secondarily associated with organic matter and oxide minerals in the Floridan Aquifer System limestones (Figure 4-12). Pyrite (an iron sulfide mineral that can also contain arsenic) occurs as single crystals and framboids less than 15μm in diameter that comprise a trace fraction of the aquifer matrix. Analyses of a substantial number of pyrite samples from various lithologic units of the aquifer by electron probe microanalysis (EPMA) suggest that arsenic is incorporated into the pyrite mineral structure (Figure 4-13). It was also determined that the leachable fraction of the total arsenic is only ~2 percent with no correlation between the leachable arsenic concentration and the total arsenic concentration in the aquifer rock.

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BOX 4-7 Continued

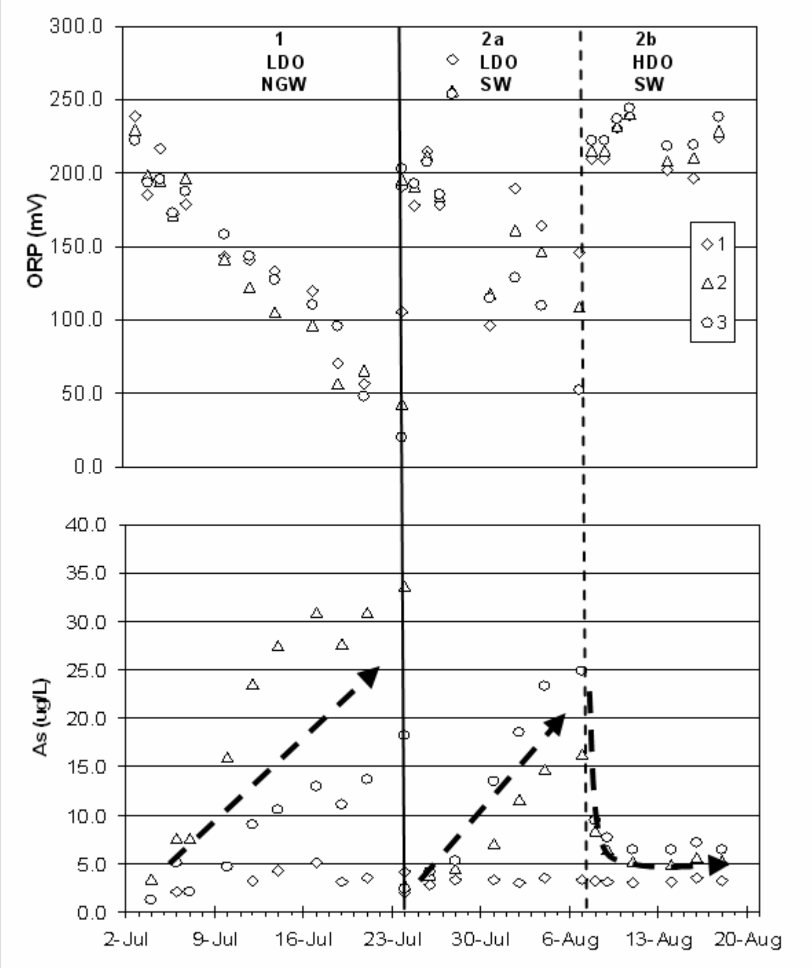

FIGURE 4-11 Results of bench-scale leaching study for three carbonate rocks exposed to ASR source water.

In summary, a combination of characterization and experimentation in the laboratory has been used to determine that desorption or oxidative dissolution is the most likely mechanism causing arsenic release from the aquifer solids to the stored water following the initial recharge event. Depending on subsequent redox conditions, arsenic is favored to be mobilized or demobilized in association with iron (although not in stoichiometric proportions). Therefore, it is hypothesized that the dynamic redox conditions expected in field ASR systems during repeated cycles of recharge, storage, and recovery may recreate conditions favoring intermittent arsenic mobility.

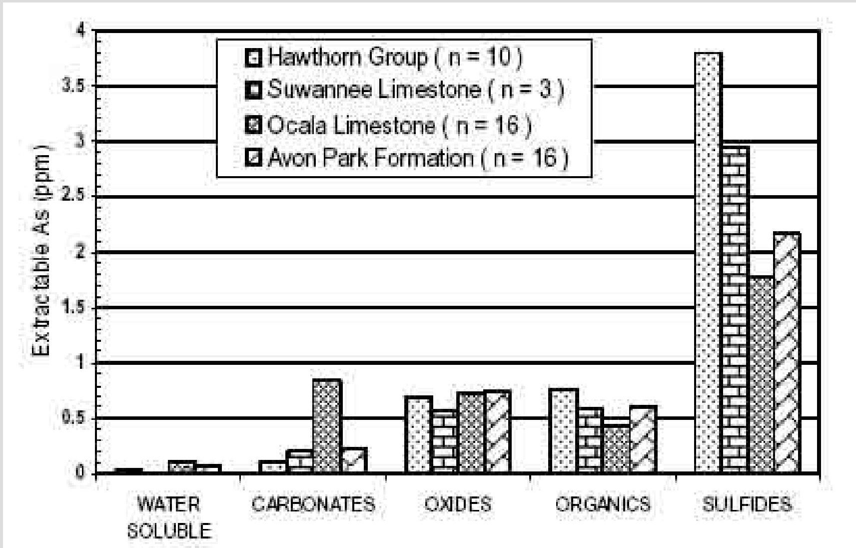

Figure 4-12 Arsenic concentrations (recalculated to parts per million) obtained from sequential extraction of four different lithostratigraphic units of the Floridan Aquifer System. SOURCE: Arthur et al. (2007).

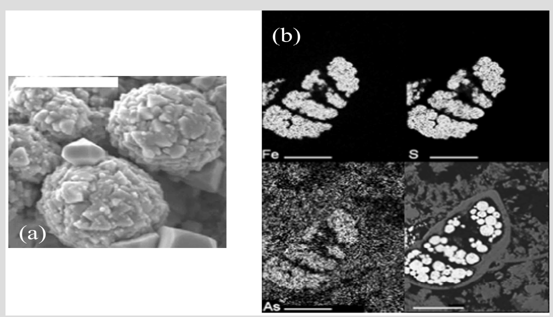

FIGURE 4-13 Direct image and element analysis demonstrating that arsenic can be associated with framboidal pyrite in Floridan aquifer limestones: (a) scanning electron microscope image (Price and Pichler, 2006); (b) electron probe microanalysis element maps and a corresponding backscatter electron image show elevated arsenic associated with framboidal pyrite (iron sulfide) clustered within a foraminifera (Arthur et al., 2007). All scale bars are white and are approximately 10 μm.

removal via filtration or ORP reduction through the use of chemical additives can be evaluated at the bench scale. Results can then be scaled-up to field applications.

Effects of Heterogeneity on Sampling and Prediction

In the bench-scale leaching example (Figure 4-11), the effects of sample heterogeneity are evident. An order-of-magnitude difference is observed in the amount of arsenic released from the core material into the leachate, reflecting chemical and mineralogical heterogeneities in the rocks. As noted above, variations exist in source water chemistry as well. If the source of the MUS system is a surface water body, water quality varies in response to numerous factors including climatic and seasonal changes, rates of nearby pumping, and spatial variations. Whether collecting rock, sediment, or water samples for use in bench or column studies, the sampling protocol should be designed to (1) minimize contamination; (2) preserve the natural condition of the sample; and (3) represent "average" conditions as well as end-member (anomalous) conditions.

Referring back to the pyrite example, ideally the cores would have been collected and stored in an oxygen-depleted environment with native groundwater in the pore space to maintain the stability of pyrite in the matrix and limit potential atmospheric oxidation. There are often times when sample availability, cost, or logistics hinder optimal sample preservation. Samples for the bench study were collected to represent the overall lithology, mineralogy, and texture comprising the proposed ASR storage zone. In an attempt to bracket the full range of heterogeneity, samples with anomalous characteristics, both "clean" carbonates (i.e., free of reduced zones, siliciclastics, pyrite, organic matter) and those suspected of high-arsenic-bearing phases were collected. Geophysical logs, petrographic or binocular descriptions, and lithogeochemical analyses help identify samples to meet these criteria. With regard to sample size, the larger the sample, the more likely it is to represent the natural system; however, this must be balanced with limitations of working at the bench scale.

Hydrogeochemical or microbiological trends and processes observed (or inferred) in laboratory experiments are intended to broadly characterize water quality changes in field or operational conditions. Several challenges exist, however, in scaling up laboratory results to field applications. Issues of volume and scale, physical aquifer characteristics (e.g. dual porosity, preferential pathways), reaction kinetics during fluid flow and storage often preclude direct transfer of bench or column study results to the field. Laboratory studies reflect relative water quality changes that may be observed in field testing or may bracket the range of hydrochemical and microbiological processes during MUS operations. Among the factors that can transfer results from lab-scale studies directly to the field are temperature, pressure, sediment compaction, lack of representative samples or conditions, redox conditions, water-rock surface area ratio, variability in source water composition, source water-groundwater mixing, effects of dual porosity, differences in microbial population diversity, and activity.

Many of these limitations are handled readily by appropriate upscaling (in either time or space) approaches. Regardless of these caveats, bench or batch and column studies provide valuable information on potential water-rock-microbial interactions in a generally cost-efficient manner. Results of these studies may be used as inputs or validation for geochemical models and, most importantly, serve as a tool to screen potential water quality issues associated with MUS.

Comprehensive Methods for Examining Water and Aquifer Media as a Precursor to Geochemical Modeling

Advances in analytical techniques have made it possible to obtain relatively inexpensive broad-spectrum chemical analyses of inorganic constituents in rock, sediment, and water samples. If more than 5-10 constituents are to be analyzed, it is often less expensive to obtain a full-spectrum analysis that utilizes multiple analytical techniques with suitable method detection limits. Among these techniques are inductively coupled plasma-mass spectrometry, instrumental neutron activation analysis, and ion chromatography. Different instruments yield optimum results depending on the analyte and type of sample, and many commercial laboratories offer analytical packages that optimize these combinations for improved accuracy and precision. From a geochemical modeling perspective, the full-spectrum cation and anion analysis of water samples is preferred to assess data quality via calculation of charge balance error. Also of importance is determination of redox conditions, which can be measured by an ORP probe or (preferably) by the dominant redox couple, such as sulfate-sulfide or ferric-ferrous iron. Complete inorganic geochemistry and physical parameters provide important context to predict and interpret geochemical reactions involving organic and inorganic constituents. The same approach applies to the analysis of aquifer solids. For example, trace metal concentrations in the parts-per-billion range can affect water quality in an MUS system due to water-rock interactions.

Additional considerations with regard to lithogeochemical or hydrogeochemical analyses are sample preparation and analytical techniques. These procedures should follow accepted industry standards, such as EPA or American Society for Testing and Materials (ASTM) methods; however, customized methods may be required for specific experiments to test a particular hypothesis. It is often required that a certified laboratory complete the analyses. Certifications include the National Environmental Laboratory Accreditation Conference (NELAC), Standards Council of Canada (SCC), and Canadian Association for Environmental Analytical Laboratories (CAEAL).

Geochemical Modeling

A model is a calculated or constructed representation with inherent uncertainty and may reflect a process or an object. With regard to the physical, chemical, and biological aspects of MUS, numerous types of models exist. Models involving water quality changes during MUS, as well as most geochemical models, are based on a conceptual model that describes the geochemical and/or hydrologic system. Other aspects of the conceptual model include whether or not it is in equilibrium and what extents are defined for the system. Once the initial system is defined, various models can be employed to reflect equilibrium reactions and reaction paths.

Bloetscher and others (2005) describe several objectives and issues that pertain to the development of an acceptable model involving aspects of groundwater injection. Among the objectives they outline that are most relevant to MUS water quality changes are (1) to predict the concentration of contaminants with time from the source to the observation points, and (2) to determine the effects of retarding factors on contamination concentration (dilution, dispersion, adsorption, time decay). Geochemical modeling objectives (modified from Mirecki, 2006), specifically with regard to ASR include characterization of (1) mixing between native groundwater and recharge water during cycle testing; (2) geochemical reactions that occur during all phases of cycle tests in different lithologies; (3) controls on fate and transport of mobilized metals during ASR cycle testing; (4) uncertainty due to the use of incomplete water quality data sets; and (5) bracketing rock and water compositions to represent natural system heterogeneity in the model.

Implicit in item 4 above is a larger concern about data quality and quantity. With regard to data quality, considerations exist in terms of sampling protocols, analytical instrumentation, methodologies, accuracy and precision, charge balance, and laboratory certification (see previous section)

Data quantity can be discussed in the context of number of samples and number of analytes. Chapter 6 discusses strategies for sample frequency and spatial distribution for MUS water monitoring. The earlier section titled "Heterogeneity Effects on Sampling and Prediction" describes the importance of collecting a range of lithologies, including various textures and mineral assemblages to represent the full range of phases and compositions. With regard to analytes, multielement-multimethod analytical packages are preferred to provide a more robust understanding of the hydrogeochemical system. Whether the samples are water or aquifer solids, sample selection is driven by the modeling objectives.

Numerous geochemical models exist, and most have overlapping capabilities. These models address the hydrogeochemical and microbiological processes outlined earlier in this chapter. Perhaps the most widely used public domain code is PHREEQC (Parkhurst and Appelo, 1999), version 2.2 of which has the following simulation capabilities: ion exchange equilibria, surface complexation equilibria, fixed-pressure gas-phase equilibria, advective transport, kinetically controlled reactions, solid-solution equilibria, fixed-volume gas-phase equilibria, variation of the number of exchange or surface sites in proportion to a mineral or kinetic reactant, diffusion or dispersion in 1D transport, 1D reactive transport, 1D transport coupled with diffusion into stagnant zones, and isotope mole balance in inverse modeling. A similarity robust code is the commercial product Geochemist's Workbench (GWB) Professional Version 6.0 (Bethke, 1996), which additionally includes 2D reactive transport modeling, heat flow, variably spaced grids, and flexible boundary conditions and provides enhanced graphics.

Additional examples of MUS-relevant geochemical models include EQ3NR (Wolery, 1992)—a code that calculates geochemical aqueous speciation and solubility, and RETRASO (Saaltink et al., 2004)—a code for modeling reactive transport of dissolved and gaseous species in variably saturated porous media, among other capabilities. These latter two codes, for example, were applied in a study of the hydrogeochemical effects of recharging oxic water into an anoxic pyrite-bearing aquifer (Saaltink et al., 2003). Another reactive transport model, TOUGHREACT (Xu et al., 2004), is a comprehensive simulator that considers numerous subsurface thermophysical-chemical processes under various thermo-hydrological and geochemical conditions of pressure, temperature, water saturation, and ionic strength. TOUGHREACT can be applied to one-, two-, or three-dimensional porous and fractured media with physical and chemical heterogeneity. Mineral dissolution-precipitation can take place subject to either local equilibrium or kinetic controls, with coupling to changes in porosity and permeability and capillary pressure in unsaturated systems. Chemical components can also be treated by linear adsorption and radioactive decay (Xu et al., 2004). PHT3D is a model that couples three-dimensional transport to a geochemical model; the robust utility of which is described in an aquifer storage transfer and recovery case study (Box 4-8, adapted from Prommer and Stuyfzand, 2005; Figure 4-14)

Easy-Leacher® (Stuyfzand, 2002) is a user-friendly 2D reactive transport code programmed within an MS EXCEL® spreadsheet for predicting water quality changes during artificial recharge (basins, recharge wells, or ASR) and river-bank filtration. The modeling code combines physical and chemical principles with empirical rules based on nearly three decades of artificial recharge experiments and studies at Kiwa Water Research, Netherlands. Parameter inputs include major water chemistry constituents, trace metals, radionuclides, organic pollutants, and pathogens. Easy-Leacher calculates water quality changes in MUS systems, including recharge basins and changes reflected in hydrochemical fronts due to aquifer matrix leaching. A few examples of processes considered in the calculations are water mixing, sulfide and organic matter oxidation, dissolution of oxide and carbonate minerals, sorption, and radioactive decay. The application also calculates sludge accumulation rates for recharge basins.

The aforementioned modeling codes comprise only a subset of those available either commercially or via public domain access. Although mention of these models in this report is not an endorsement thereof, examples are provided to illustrate the broad scope of applications for geochemical models, focusing

BOX 4-8
Geochemical Modeling of an Aquifer Storage Transfer and Recovery (ASTR) Facility Using PHT3D

- Setting: Study of an aquifer storage transfer and recovery (ASTR) project was completed in the Netherlands to assess the technical feasibility of utilizing deep-well direct-aquifer recharge of canal water to offset water table drawdown and restore local wetlands with recovered water. The pilot plant was constructed along the canal bank near Someren, southern Netherlands, and includes an intake, a pretreatment facility, a recharge well, four monitoring wells and a recovery well. The recovery well is located 98 m west of the recharge well, and two monitoring wells are located along that flowpath: 8 and 38 m west of the recharge well. The other two monitoring wells are located 12 and 22 m east of the recharge well. Pretreatment is comprised of flocculation, flotation and sand filtration. Recovered water is discharged to a storage pond and ultimately returns to the canal. The aquifer system is siliciclastic, with clay and fine-sand low-permeability interbeds separating up to four aquifers, labeled A/B, C, D, and E. The screened intervals for the recharge and recovery wells extend approximately from 280 to 310 m below land surface and 278 to 298 m below land surface, respectively. Recharge extended 854 days at a rate of 720 m^3 from day 0 to 726, and 960 m^3/day from day 727 to 854.
- Data collected: Sediment cores were collected and preserved for geochemical analysis and sequential extraction. Core preservation included on-site sealing in liquid paraffin and storage at 4°C in the dark. Samples were pretreated in an anoxic glove box. Water quality parameters, piezometer readings, and temperature (as depth profiles) were recorded over the 854-day recharge event.
- Numerical model: PHT3D is a three-dimensional advective-dispersive multicomponent reactive transport model used in this study. The model couples a three-dimensional transport simulator (MT3DMS; Zheng and Wang, 1999) with the geochemical model PHREEQC-2 (Parkhurst and Appelo, 1999). The combined functionality of these models, through PHT3D, allows robust assessment of non-reactive and reactive transport, heat transfer, equilibrium, and kinetic reactions, as well as redox, ion exchange, and precipitation-dissolution processes.
- Conceptual model: Regional groundwater flow was a negligible component of the three-dimensional flow field. As such, the symmetrical flow field was represented along the flow path by a half-model of 263 m and 124 m perpendicular to flow and symmetry axis. Boundary conditions were established as no-flow parallel to the main flow direction and fixed-head, fixed concentration along the perpendicular axis. The model was discretized into 12 layers of variable hydraulic conductivities based on a previous study. The hydraulic conductivity distribution was slightly modified during model calibration.
- Model domain components and calibration: These components of the model can broadly be described as nonreactive and reactive transport, kinetic controls, and modeled source and groundwater compositions. Owing to its contrast between the source and groundwater, chloride served as a suitable tracer for calibration of the nonreactive transport component of the model. Variability in chloride concentrations in the injected water was addressed by discretizing the 854-day simulation into 39 stress periods. A subset of the temperature-depth profile data constrained the heat transport model, which not only improved calibration of the hydraulic conductivity distribution, but allowed for characterization of spatial and temporal variation as it relates to rates of temperature-dependent chemical reactions.

The reaction network, on which reactive transport simulations were based, included the following:

- Equilibrium-based speciation and redox reactions of all major ions
- Cation exchange and equilibrium for ferrihydrite (an HFO)
- Sediment-bound organic matter in its role as a source of dissolved organic carbon
- Kinetic reactions including pyrite oxidation via oxygen and nitrate, and conversion of organic to inorganic carbon (DOC mineralization) via nitrate, oxygen, and sulfate.

Despite mineralogical heterogeneity in the aquifer matrix, ambient groundwater compositions were found to be fairly homogeneous; thus the initial concentration was based on the hydrochemistry of one representative sample. Temporal variations in source water composition were reflected in the model; and both native groundwater and seasonal recharge waters were charge-balanced through minimal (<5 percent) adjustment of the chloride concentration. Model calibration was based primarily on equilibrium reactions and DOC mineralization and pyrite oxidation kinetics. Pyrite oxidation was found to account for a large proportion of the oxygen and nitrate removal; cation exchange reactions were found to have only a minor impact on the simulated versus observed breakthrough curves.

- Synopsis of results: Observed and calibrated model chloride concentrations and temperature variations at most locations were satisfactorily reproduced. For some parameters, breakthrough curves were substantially affected by reactive processes and others were not, depending on the well from which the observational data were collected. Seasonal variations are clearly observed in the data and are reflected in model results. Observed and simulated oxygen and nitrate concentrations are shown in comparison with model runs that were fixed in time and space at 8, 14, and 20^{o}C (Figure 4-14). The model run reflecting the simulated temperature field for calculation of reaction rates yielded the best match (T = variable). Aquifer physical and chemical heterogeneity also had a significant effect on the location and rate of removal of oxygen and nitrate; simulated and observed breakthrough curves were well matched by the model (not shown). Comparison of simulated results with observational data indicated that despite the heterogeneity of the system, coupled reaction and transport that occurred during the experiment are well characterized. This good agreement is attributed to the detail in which the hydrogeological and hydrochemical aquifer characterization was completed, as well as incorporation of the temperature dependence of reaction rates.

continues next page

BOX 4-8 Continued

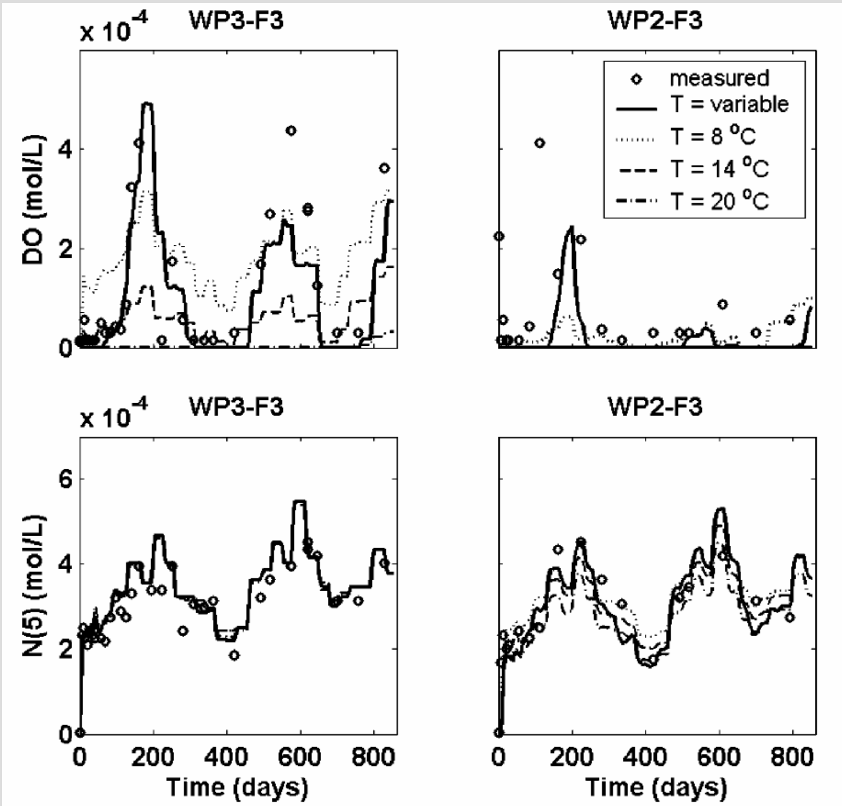

FIGURE 4-14 Measured and simulated breakthrough curves of oxygen and nitrate at 8-m (WP3) and 12-m (WP2) distances from the recharge well in aquifer zone D (after Prommer and Stuyfzand, 2005). Adapted from Prommer and Stuyfzand (2005). Copyright 2005 by American Chemical Society.

SOURCE: From Prommer and Stuyfzand (2005) and supporting documents online at *http://pub.acs.org.*

primarily on inorganic constituents. Models for organic and microbiological processes are also widely available.

CONCLUSIONS AND RECOMMENDATIONS

Conclusion: There is a substantial body of work documenting improvements in water quality that can occur in an MUS system, particularly those that involve surface spreading. The subsurface has, to a greater or lesser extent, the capacity to attenuate many chemical constituents and pathogens via physical (e.g., filtration and sorption), chemical, and biological processes. In places where the groundwater quality is saline or otherwise poor, the implementation of MUS will likely improve overall groundwater quality and provide a benefit to the aquifer.

However, the type of source water used for recharge along with subsurface properties and conditions influences the extent of treatment and the effects on native groundwater quality. Therefore, a thorough knowledge of the source water chemistry and mineralogy of the aquifer is requisite to embarking on any MUS project. It is important to establish whether the mixing of source water and native groundwater, as well as chemical interaction with aquifer materials, yields compatible and acceptable effects on water quality.

Recommendation: A thorough program of aquifer and source water sampling, combined with geochemical modeling, is needed for any MUS system to understand and predict the medium- and long-term chemical behavior and help determine the safety and reliability of the system.

Conclusion: A better understanding of the contaminants that might be present in each of the potential sources of recharge water is needed, especially for underutilized sources of water for MUS, such as stormwater runoff from residential areas. Limited data exist on the use of urban stormwater for MUS systems. Consistent with an earlier National Research Council report (NRC, 1994), urban stormwater quality is highly variable and caution is needed in determining that the water is of acceptable quality for recharge.

Recommendation: Research should be conducted to evaluate the variability of chemical and microbial constituents in urban stormwater and their behavior during infiltration and subsurface storage to establish the suitability of combining MUS with stormwater runoff.

Conclusion: The presence and behavior of emerging contaminants (e.g., endocrine disrupting compounds, pharmaceuticals, and personal care products) is of concern, especially with reclaimed wastewater. However, the concern about these compounds is not unique to MUS systems. Surface waters and groundwaters around the nation carry the same kinds of chemicals, and surface water treatment systems are not normally designed to address them.

Recommendation: Basic and applied research on emerging contaminants that has begun at a national scale should be encouraged, and MUS programs will be among the many beneficiaries of such investigations.

Conclusion: A better understanding is needed of potential removal processes for microbes and contaminants in the different types of aquifer systems being considered for MUS. These studies need to assess spatial and temporal behavior during operation of an MUS system. This research will reduce the uncertainty regarding the extent of chemical and microbial removal in MUS systems. In addition, this information will help reduce impediments to public acceptance of a wide variety of source waters for MUS.

Conclusion: In particular, changes in reduction-oxidation (redox) conditions in the subsurface are common and often important outcomes of MUS operation. These changes can have both positive and negative influences on the physical properties and the chemical and biological reactivity of aquifer materials. For example, the existence of both oxidizing and reducing conditions might enhance the biodegradation of a suite of trace organic compounds of concern or, conversely, lead to accumulation of an intermediate product of concern. Redox changes can cause dissolution-precipitation or sorption-desorption reactions that lead to adverse impacts on water quality or clogging of the aquifer; however, such precipitation reactions can also sequester dissolved contaminants.

Recommendation: Additional research should be conducted to understand potential removal processes for various contaminants and microbes and, particularly, to determine how changes in redox conditions influence the movement and reactions for many inorganic and organic constituents. Specific areas of research that are recommended include (1) bench-scale and pilot studies along with geochemical modeling to address potential changes in water quality with variable physical water conditions (pH, Eh, and DO); and (2) examination of the influence of sequential aerobic and anaerobic conditions or alternating oxidizing and reducing conditions on the behavior of trace organic compounds in MUS systems, especially during storage zone conditioning.

Conclusion: Molecular biology methods have the potential for rapid identification of pathogens in water supplies. These noncultivable techniques have not been tested in a meaningful way to address background and significance of the findings. False negatives and false positives remain an issue that needs to be addressed.

Recommendation: Research should be conducted to address the approaches and specific applicability of molecular biology methods for pathogen identification, particularly interpretation of results that cannot determine viability, for the different types of source waters and aquifer systems being considered for MUS.

Conclusion: Pathogen removal or disinfection is often required prior to storing water underground. If primary disinfection is achieved via chlorination, disinfection by-products such as trihalomethanes and haloacetic acids are formed. These have been observed to persist in some MUS systems. However, chlorine is the most cost-effective agent for control of biofouling in recharge wells; hence, it may not be possible to eliminate entirely the use of chlorine in MUS systems (e.g., periodic pulses of chlorine to maintain injection rates).

Recommendation: To minimize formation of halogenated DBPs, alternatives to chlorination should be considered to meet *primary* disinfection requirements, such as ultraviolet, ozone, or membrane filtration.

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5
Legal, Economic, and Other Institutional Considerations

Is managed underground storage of recoverable water (MUS) being utilized in circumstances where it is appropriate, given costs and environmental concerns, or do institutional barriers impede its use? How are regulatory agencies, courts, and other institutions involved with the development and oversight of MUS facilities? Does this involvement support the safe, efficient, and cost-effective use of MUS technologies, with maximum benefits and minimum costs, balancing the interests of the project proponents, society, and the environment?

These questions are critical ones, because MUS has been studied for decades in the water resource management literature and has been successfully implemented by multiple jurisdictions. Although the previous chapters have described the physical challenges associated with MUS, those challenges are not the only impediments to its more widespread implementation. An equal or greater challenge, and the topic of this chapter, is the array of institutional issues associated with MUS.

MUS technologies have been applied in a wide range of physical systems (e.g., different aquifer types, different hydrogeological and geochemical conditions, and different depths) and for a wide range of purposes (municipal water supply, agricultural and industrial water supply, and even supplies for aquatic habitat) and operational goals (peak and seasonal demands, drought and other emergency supply). As the applications and understanding of MUS to meet different water management goals and water supply needs increase, and the ability to meet technical challenges associated with these technologies improves, MUS is increasingly being considered and applied throughout the United States.

The decision to utilize MUS will reflect both technical and institutional considerations. As the technical challenges associated with MUS become more tractable, the institutional issues associated with its implementation rise to equal or even greater prominence. "Institutional issues" refer to topics associated with governance, informed decision making, legal rights and liabilities, economic trade-offs under uncertainty, and so on. As others have recognized, institutions are key elements of water resource management (Blomquist et al., 2004; Ingram et al., 1984; Livingston, 1993; Lord, 1984).

At the outset, it should be noted that MUS is likely to be utilized only when it is less costly than alternative means of meeting water demand. As discussed in this chapter, although economic studies have been performed on various aspects of MUS (e.g., the economics of groundwater use or of artificial recharge), little has been published in terms of formal studies of the economics of MUS versus other forms of water storage and water management. Consequently, references to MUS as "costly" or "inexpensive" are usually generalities. Whether

MUS is economically feasible depends on the circumstances of particular locations—not only the technical requirements of a particular MUS project, but the alternatives that are available for water supply and storage and the financial resources that can be marshaled.

Municipal and industrial suppliers in water-short regions, for example, are able to pay almost any price to meet water demands that are increasing in the face of growing populations or to respond to the mining of groundwater aquifers, increasing regulatory constraints on surface water storage, and regional water competition. Furthermore, communities in almost any location have alternative means of addressing these water demands, such as conservation measures, pricing practices, or transfers of water from other uses (e.g., retiring of agricultural water rights is occurring across the western United States).

Institutional arrangements also determine whether MUS comes within the set of feasible policy options. Institutional constraints affect whether recovered water can be stored underground, that is, whether a legal regime exists that would prohibit or permit this activity. The coordinated actions necessary for implementation of an MUS program are unlikely to occur if rules and organizational arrangements (1) impede or prohibit coordination of actions necessary to divert, impound, treat, recharge, store, protect, and extract water; (2) do not protect those who invest in facilities or who store water now for later recovery; or (3) do not provide or recognize workable and fair methods for distributing the costs of an MUS program among those who benefit from it (Blomquist et al., 2004).

Those who would invest in MUS projects need to capture and internalize benefits from their investments. Those who incur costs by participating in an MUS program (e.g., accepting recovered water supplies in lieu of other supply sources to which they also have access) must be able to capture some of the benefits they have provided for others. The assurance of the protection of public health and the environment is also critical in MUS development and operation.

Other major institutional considerations in MUS involve the nature of the organizations (public or private) and the allocation of their authority and responsibility to capture, convey, manage, store, or sell water; to monitor water resource conditions and respond to perceived problems; to communicate with the public and other policy makers; and to protect public interests. Like any approach to water management, MUS emerges through the interaction of multiple organizations with diverse interests and responsibilities. The practices of those organizations and the relationships between them shape the implementation and performance of MUS. This chapter provides an overview of the regulatory involvement in the development and oversight of these technologies; a discussion of other issues facing institutions in their approach to MUS; and an evaluation of the economic aspects of MUS.

LAW, REGULATIONS, AND THE MANAGED UNDERGROUND STORAGE OF RECOVERABLE WATER

At each step in the development and implementation of a water storage and delivery project there are institutional issues to address. One of the reasons for the complexity of the development of MUS systems is that the action of taking water, placing it in storage through a well or recharge basin, storing it underground in an aquifer, and removing it from the aquifer (typically through a well) for later use—particularly if that use is for drinking water—involves a range of regulatory programs at the federal, state, and sometimes, local levels. MUS projects are among the most complex to implement, unless a state has addressed these issues in a statutory scheme that was created specifically for the regulation of these projects. Box 5-1 delineates the aspects of MUS activities that may be subject to regulatory oversight.

Recharge and recovery projects involve an array of legal issues. Depending on a state's laws and regulations, MUS projects will be easier or more difficult to develop and implement. States' legal regimes governing water are infamous for separating water allocation or rights issues from those of water quality. The fundamental concerns of water quantity and water quality laws are usually quite distinct, as are the agencies that administer these laws. Statutory schemes that are specifically directed at MUS projects contain a welcome recognition that these different aspects of water are interrelated and appropriately considered in tandem. While some states have comprehensive regulatory schemes, others have schemes developed for different types of quality concerns or very minimal systems. Any discussion of water quality protection is further complicated because both the federal and the state governments play roles in regulation. Laws allocating water quantities among uses and users are discussed in the following subsection, followed by a discussion of water quality concerns.[1]

MUS and the Regulation of Water Use

Well-understood and characterized rights of water use are essential for MUS projects to be considered feasible options for water management. Most states' water rights systems were developed long before groundwater storage was contemplated. Additionally, competing rights holders will be vigilant to prevent infringement of their rights and will be involved in any proposals that are perceived to affect their water.

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[1] A very useful review of laws and regulations concerning the aquifer storage and recovery method of MUS was provided by Seerley (2003).

BOX 5-1
Aspects of MUS Activities That *May* Be Overseen by a Regulatory Agency, Depending on laws or regulations applicable at the site

Water Quantity-Related Activities

- The right or permission to store water within an aquifer, the volume of water that can be stored, and the protection of the stored water from recovery by others)
- The timing and rate at which stored water can be recharged to the aquifer to prevent impacts to subsurface structures from mounding of water levels or stream accretions resulting from recharge
- The right or permission to withdraw the water from storage (this can be particularly important in regions where groundwater management or groundwater recovery activities are restricted due to water quantity-related concerns such as falling groundwater levels, land subsidence, or saltwater intrusion)
- The timing and rate at which stored water can be recovered to prevent water quantity-related aquifer management concerns, such as well interference or other impacts of neighboring well users, and stream depletions or other surface water impacts for tributary aquifers
- The type of use to which the recovered water can be put

Water Quality-Related Activities

- Protection of the quality of the native water in the aquifer from impacts by or degradation from interactions with the water to be recharged; if recharge is by well injection, this is typically regulated under the federal Underground Injection Control program
- Protection of the quality of the water being stored from impacts by or degradation from interactions with the surrounding native water in the storage aquifer, particularly if the intended post-recovery use of the stored water is for potable purposes
- Protection of the aquifer matrix from physical impacts resulting from chemical interactions between the stored and native waters, such as precipitation of metals and resultant clogging of aquifer pore spaces (this can also be viewed as a water quantity-related issue, and regulated by a water resources agency because these impacts can reduce aquifer productivity for other well users)
- The construction and maintenance of wells, including well casing and wellhead, to prevent movement of water between aquifers and water and to prevent contaminants from entering the aquifer unintentionally
- The construction and maintenance of surface recharge facilities

Land Use

- Ownership of and/or access to land for surface recharge
- Ownership of and/or access to land for well installation, operation. and maintenance, for directionally drilled recharge or dual-purpose recharge and recovery wells, this may also include ownership of land over the entire length of the well
- Ownership of and/or access to and permission to use the storage aquifer;

In addition, special laws or regulatory programs may address the water quantity and/or water quality aspects of activities involving recycled wastewater, stormwater, desalinized water, or other forms of water reuse.

Surface Water and Groundwater Rights

One set of water rights issues arises out of the presence of dual or multiple water rights systems, which separate the management of surface and groundwater. Separate rules governing surface water and groundwater are common throughout the United States, although the rules in use differ noticeably between the eastern and western states.

In the United States, most states east of the Mississippi River provide riparian rights for the use of surface water; that is, they link the use of water to the ownership of land adjacent to that body of water. Another set of rules governs groundwater use rights—by virtue of their land ownership, overlying owners have correlative rights to withdraw water from beneath the land for beneficial uses on the land. Water shortages (relatively rare in the East through most of the nineteenth and twentieth centuries) occasionally caused one landowner's water use to encroach upon the needs or customary use of another, and these were generally approached through common law remedies. During the latter half of the twentieth century and into the early twenty-first century, eastern states have modified their water rights regimes by requiring state-issued permits limiting water withdrawals to a maximum quantity or rate (e.g., gallons per minute or per day). Furthermore, all eastern states overlying the aquifers of the Coastal Plain—from New Jersey south to Florida—have enacted special regulatory programs for use in designated locations (which may be called "Capacity Use Areas," "Critical Areas," or "Groundwater Management Areas") where groundwater resources have been overdrafted or where negative impacts such as well interference, seawater intrusion, or land subsidence have necessitated a more active regulatory and regional approach. The legal context for MUS projects in the eastern states is thus comprised of the overlaying of permit systems and critical area designations on the existing riparian rights rules for surface water and correlative rights rules for groundwater.

This is of special significance because most MUS projects that have been planned or undertaken along the eastern seaboard of the United States are in the Coastal Plain, where these state-by-state regulatory programs apply. Some of these regulatory regimes include strict limitations on groundwater use in state-designated critical areas and may require consideration of drawdown impacts of one pumper on others within the same area. Often, these regulatory programs restrict withdrawals from designated aquifers, but allow the use of MUS to provide "credits" that project proponents can draw against.

Most western states in the United States developed rights to the use of surface waters by means of the prior appropriation doctrine. The prior appropriation doctrine allocates water on the basis of seniority, or "first in time, first in right," rather than on the basis of land ownership Through agency-issued permits or a process of adjudication, individuals are granted rights to divert from the stream channel and use up to a specific amount of water, usually on an annual basis. When shortages occur, those who hold the most senior rights have those rights satisfied first, while those who hold junior rights may not receive any water.

States were slower to develop statutory schemes to address the exploitation of groundwater, because it was only with the widespread utilization of pumping that conflicts began to arise. Some states regulated groundwater through the prior appropriation doctrine, requiring permits for withdrawal and protecting other users from excessive withdrawals. Other states permitted landowners unlimited access to the resource. States have also regulated groundwater on a regional scale, through critical area designations or similar means, with more stringent controls in some regions than others. As groundwater is better understood and the competition for water increases, there is increasing regulation by states.

MUS projects typically involve the movement of surface water into groundwater and thus there is a need to reconcile legal systems that typically do not integrate these differing concerns. In states where rights for use of surface water differ from rights for use of groundwater, some adjustment of water rights rules may be necessary for the holder of a surface water right to be able to legally store some of that water underground and pump it out later. By the same token, the rights of a groundwater user to put water into an aquifer, as well as take it out later, may require modification of governing rules.

For instance, if an individual or organization already possessing rights to the use of groundwater also participates in an MUS project, the project proponent will have to establish how the stored water relates to the rights holder's other groundwater extractions—that is whether stored water is counted as the "first" water extracted (after which the rights holder can continue to extract whatever other amount of groundwater it has a right to use) or as the "last" water extracted (in which case a rights holder does not tap its stored water in a given time period unless and until it has already extracted whatever other groundwater it had a right to use) (Shrier, 2004). The implications of the difference are considerable. The former option provides little incentive for the holder of an existing groundwater right to engage in long-term water storage since the stored-water "account" is exhausted first. The latter option provides a considerable incentive to store water for the long term, but may not account for the benefits to other aquifer users that accrue when a rights holder places water into the aquifer and leaves it there for a long period (discussed later in this chapter).

Storage and Recovery of Project Water

Another set of legal concerns is raised because many MUS projects involve the storage of water imported from another location or produced through purification processes (e.g., reclaimed wastewater, desalinated ocean or brackish water). In most states this "project water" is produced and delivered by public or private project operators and does not fall clearly within the riparian, appropriative, or other rights systems that apply to surface water diversions or groundwater extractions. Contracts between project operators and the recipients of the project water express rights in the water. These contracts come in such variety that it is difficult to characterize a typical arrangement.

Legal Status of Aquifer Storage Space

A third major legal issue is unique to underground storage projects and presents novel questions. While ownership of groundwater rights has been developed in western states, there is no readily available reference for ownership or control of aquifer storage rights. Thus, in the absence of a statutory provision, it is often unclear whether aquifer space is owned or controlled by overlying property owners, by owners of water use rights in the aquifer, or by no one at all.

In some states, this issue has been addressed by statutory and regulatory schemes providing for MUS, or by court decisions resolving other issues.[2] In 1995, the State of Oregon adopted a statute authorizing the state's Water Resources Commission to issue permits for aquifer injection and storage projects, and providing for the state's departments of Environmental Quality and Human Services to offer comments during the permit review process.[3] The statute imposes water quality standards on the stored water and acknowledges that the water will be retrieved sometime in the future. The Oregon statute does not require that aquifer storage and recovery projects have discharge permits,[4] and declares that water stored in ASR projects will not be considered a waste, contaminant, or pollutant.[5]

Idaho established through legislative action that the storage of water is a beneficial use, and that permits can be issued for the capture and storage of unappropriated water, in effect creating a secondary water right.[6] Idaho's approach recognizes that such projects may simply recharge groundwater supplies, whereas Oregon's approach mandates that water would be retrieved from the aquifer.[7]

In 2005 the Kansas Division of Water Resources promulgated regulations to establish a permitting process for ASR projects.[8] Project applicants must seek and obtain two types of appropriation permits. The first permit is for appropriating the surface water that will be stored underground. The second permit is for

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[2] California, for example, does not have a statewide approach to groundwater storage, but rights to store water underground and recover it later have been established through adjudications of pumping rights in several groundwater basins (Bachman et al. 1997; Blomquist, 1992; Blomquist et., 2004; Littleworth and Garner).
[3] Or. Rev. Stat. § 537.534 (2003).
[4] Or. Rev. Stat. § 537.532(b) (2003).
[5] Or. Rev. Stat. § 537.532(a) (2003).
[6] Idaho Code Ann. § 42-234(2) (2006).
[7] Idaho Code Ann. § 42-234(1) (2006).
[8] Kan. Admin. Regs. § § 5-12-1 *et seq*.

appropriating the stored groundwater—extracting it for use. The Kansas Division of Water Resources was prompted to enact these new regulations by a demonstration project in the Equus Beds groundwater area of the Little Arkansas River in south-central Kansas. Wichita and the Equus Beds Groundwater Management District No. 2 are undertaking the ASR project, with the city as the designated lead local agency (Peck and Rolfs, 2005).

Arizona has enacted a comprehensive statute addressing the storage of water. Arizona Revised Statutes § § 45-801.01 *et seq.* has a twofold purpose:

1. Protect the general economy and welfare of this state by encouraging the use of renewable water supplies, particularly the state's entitlement to Colorado River water, instead of groundwater through a flexible and effective regulatory program for the underground storage, savings and replenishment of water.
2. Allow for the efficient and cost-effective management of water supplies by allowing the use of storage facilities for filtration and distribution of surface water instead of constructing surface water treatment plants and pipeline distribution systems.[9]

The storage facilities cannot impair vested water rights, and the applicant for a water storage permit must have a right to the proposed source of water.[10]

Unlike Oregon, Idaho, and Arizona, California does not have a comprehensive act for the underground storage of water. This is in part due to California's common law treatment of water rights in which a property owner has the right to the surface and everything above or below it. Therefore, storage could be detrimental to an overlying property owner's right.[11] However, California does recognize the underground storage of water as beneficial use, as depicted in California Water Code, Section 1242:

> The storing of water underground, including the diversion of streams and the flowing of water on lands necessary to the accomplishment of such storage, constitutes a beneficial use of water if the water so stored is thereafter applied to the beneficial purposes for which the appropriation for storage was made.[12]

Texas also uses a common law approach, molded after the Rule of Capture and its treatment of oil and natural gas.[13] However, the Texas Water Code contains a preliminary regulatory scheme that proposes the investigation of aquifer storage through the issuance of temporary permits for pilot projects: "(a) The commission shall investigate the feasibility of storing appropriated water in

[9] Ariz. Rev. Stat. § 45-801.01 (2005).
[10] Ariz. Rev. Stat. § 45-803-01(A) (2005); Ariz. Rev. Stat. § 45-831-01(B) (2005).
[11] Kiel and Thomas,2003.
[12] Cal. Water Code § 1242 (2006).
[13] Drummond et al., 2004.

various types of aquifers around the state by encouraging the issuance of temporary or term permits for demonstration projects for the storage of appropriated water for subsequent retrieval and beneficial use."[14]

As these examples and the discussion in the preceding subsections indicate, MUS projects are likely to be governed and affected by a combination of laws in each state, since MUS can involve the use of surface water or other project waters for recharge, the extraction and use of groundwater upon recovery, and the storage of water in the aquifer. A particular project can therefore require permits or other regulatory approval from multiple state agencies enforcing different provisions of state law (not to mention federal approval for injection projects, discussed in greater detail later in this chapter). It may not be necessary to rewrite state water codes in order to facilitate underground water storage, but state policy makers considering the promotion of underground storage are well advised to review current state regulatory requirements and processes in order to assess the extent to which they inhibit the planning, economic feasibility, and practical execution of MUS projects. Several states (Arizona, Colorado, Kansas, Nevada, New Mexico, Oregon, Utah, and Washington) have already modified statutes or regulations to provide for alternative permitting processes for MUS projects or to clarify the water rights aspects of underground storage and recovery of water (Shrier, 2004).

Additional Considerations

Thus, a variety of water rights issues may be triggered by an MUS proposal, with important implications for the prospects of implementing such a plan. When water rights are unquantified or otherwise incompletely specified, or aquifer storage rights are unclear, users are less likely to undertake investments in storing water or to exercise restraint in leaving stored water underground. In addition, when water rights are unclear or when differing and contestable claims arise in relation to the same water resource, users bear the additional costs of resolving conflicts and negotiating and/or enforcing solutions about who may do what in relation to which aspects of the resource. Rights to manage stored water, to exclude others from capturing it, or to transfer stored water to others help assure participants that they will maintain control of the water supplies they commit to an MUS project and, thus, be able to recover benefits from the project. Here too, however, the details of these legal arrangements matter. For example, in an appropriative rights system, the priority date of stored water may be later than (or "junior" to) that of other water rights holders in the aquifer. If junior users' rights are subordinated during periods of shortage, such an arrangement would provide no incentive to store water for water-short years.

[14] Tex. Water Code Ann. § 11.153 (2005).

Rules governing water use can have yet another effect on MUS projects. An important advantage of MUS is flexibility in the use of water. Traditional approaches to the allocation of water rights may undermine the flexibility of an MUS project, which treats as interchangeable water derived from alternative sources and withdrawn at times that cannot be specified in advance. The latter point is critically important: even in states where water use rights are quantified and limited, they may be fixed by time period (e.g., a right to use *x* amount of water per year). The recovery aspect of an MUS project cannot always be so readily fixed—stored water might be drawn on every year at a predictable rate (more likely in the event of an MUS project that is intended to augment supplies using purified wastewater) or might be drawn on only occasionally in response to drought or other interruptions of usual water supply. In the latter type of case, how much groundwater will be extracted and when are necessarily uncertain. Thus, in the same aquifer, some entities may have quantified annual rights of withdrawal while others possess a recognized yet unspecifiable right of withdrawal. The emergence and development of MUS in the United States depends therefore not only on whether states define rights that are secure enough to induce individuals to invest in MUS, but also on the ability of institutions to provide some flexibility in using water from different sources and at uneven and not entirely predictable times.

Regulation of Public Health and Environmental Concerns

MUS systems involve public health and environmental concerns on two levels: impacts to the water being stored and impacts to the water in the storage aquifer. If water is being stored for recovery for potable uses, upon recovery the water will be regulated under various federal or state drinking water protection programs. Notably, there may be little difference between the regulatory approaches to water recovered from underground storage and water recovered from aboveground storage.

A greater regulatory emphasis has been placed on the second category of concerns: the impact of the stored water on the aquifer. This is the case if the aquifer being used for storage is defined as a current or future underground source of drinking water (USDW)—generally, groundwater with a total dissolved solids (TDS) content of less than 10,000 mg/L—and if the water is being stored in the aquifer by means of injection.[15] Injection systems are regulated under the federal Safe Drinking Water Act's (SDWA's) Underground Injection Control (UIC) Program or similar state programs.

[15] There is no federal regulation of aquifer recharge using surface infiltration, although state regulations and/or federal source water protection regulations may apply.

Federal and State Underground Injection Control Regulations

Federal regulation of MUS projects covers those projects that fall under the UIC program. In accordance with the mandate of the Safe Drinking Water Act (SDWA), UIC regulations provide that "no injection shall be authorized by permit or rule if it results in the movement of fluid containing any contaminant into Underground Sources of Drinking Water, if the presence of that contaminant may cause a violation of any primary drinking water regulation under 40 CFR part 141 or may adversely affect the health of persons."[16]

The U.S. Environmental Protection Agency's (EPA's) UIC regulations classify injection wells into five categories. Injection wells that are used for MUS systems are classified as "Class V" wells because they do not fit into Classes I-IV. Examples of Class V wells cited in a 1999 EPA study included agricultural drainage wells, stormwater drainage wells, large-capacity septic systems, sewage treatment effluent wells, aquifer remediation wells, car wash and laundromat effluent wells, saltwater intrusion barrier wells, aquifer recharge and ASR wells, subsidence control wells, and industrial wells (USEPA, 1999). Thus, although most UIC-regulated wells are intended for waste disposal,[17] UIC regulations also apply to wells that are used to replenish water in an aquifer (including ASR wells).

The UIC program was developed to prevent endangerment of drinking water supplies, as explained in Section 1421 (d)(2) of the Safe Drinking Water Act: "Underground injection endangers drinking water sources if such injection may result in the presence in underground water which supplies or can reasonably be expected to supply any public water system of any contaminant, and if the presence of such contaminant may result in such system's not complying with any national primary drinking water regulation or may otherwise adversely affect the health of persons."

The implementing regulations put the burden of proof on the applicant to demonstrate compliance:

> **40 CFR 144.12(a):** No owner or operator shall construct, operate, maintain, convert, plug, abandon, or conduct any other injection activity in a manner that allows the movement of fluid containing any contaminant into underground sources of drinking water, if the presence of that contaminant may cause a violation of any primary drinking water regulation under 40 CFR part 142 or may otherwise

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[16] Aquifers that are not underground sources of drinking water are not exempted aquifers. They simply are not subject to the special protection afforded USDWs.

[17] Waste disposal appears to have been the principal regulatory concern of the federal UIC program. In its explanation of the purpose for the UIC program, the EPA web site states that "when wells are properly sited, constructed, and operated, underground injection is an effective and environmentally safe method to dispose of wastes" (*http://www.epa.gov/safewater/uic/whatis.html*; accessed March 30, 2007). Furthermore, the agencies that administer UIC regulations typically regulate many times more wells intended for waste disposal than MUS wells.

adversely affect the health of persons. The applicant for a permit shall have the burden of showing that the requirements of this paragraph are met.

Underground injection control regulations may be implemented directly by the federal government through EPA regional offices or by a state agency in states that have been granted "primacy" status for this program. In states with primacy, state regulations must be at least as restrictive as federal UIC regulations (and may be more restrictive); states must have the enforcement capacity to implement the regulations; and state regulations must be submitted to the EPA.). Florida's UIC regulations are listed in Box 5-2 as an example.

Differences of approach among EPA regions can have impacts on state-level efforts to implement MUS programs, because EPA regions have taken different positions on the issue of the proper "point of compliance" for assessing aquifer water quality. (The point-of-compliance question is discussed further below[18].) Consistency among EPA regional offices on SDWA-UIC interpretation would reduce uncertainty for decision makers in assessing the costs and benefits of an MUS project compared to alternatives such as surface storage.

Florida's regulators have interpreted SDWA language to mean that "the injection practice cannot require a public water system to have to provide a greater degree of treatment because of an injection activity than it would if the injection activity were not present. This provides some leeway from the strict interpretation that there can be no violation of a primary drinking water standard." (Florida Department of Environmental Protection, 2006)

Although the UIC program and the Safe Drinking Water Act provide a national framework for regulating the quality of water introduced directly to drinking water source aquifers, UIC and other groundwater protection programs can and do vary from state to state in their structure and in their application to recharge projects. EPA regional offices may vary in their approach to application of UIC regulations to MUS projects, and the potential risks are evaluated under differing site-specific scenarios. In some states, the state groundwater protection program may have larger budgets and staffs or more direct experience with MUS than does the EPA regional office. Nationally, the EPA has more funds available for research and the development of science-based guidelines. However, perhaps recognizing the variety of circumstances in which MUS systems have been used, the EPA has not yet developed national guidelines for MUS systems. (It is studying ASR through a national workgroup to determine whether further national direction or guidance is needed.)

Despite being authorized by the SDWA, EPA's involvement in well recharge projects raises federalism concerns that are familiar to many areas of environmental regulation. These concerns are highlighted in the MUS field

[18] With particular regard to ASR systems, Pyne (2006, pp. 393-395) has offered a list of recommendations for state regulatory programs.

BOX 5-2
Florida UIC Rules

62-528.605 Well Construction Standards for Class V Wells:

(3) Class V wells shall be constructed so that their intended use does not violate the water quality standards of Chapter 62-520, F.A.C., at the point of discharge, except where specifically allowed in Rule 62-522.300(2), F.A.C., provided that the drinking water standards of 40 C.F.R. pt. 142 (1994) are met at the point of discharge for projects and facilities described in Rule 62-522.300(2)(a) and (b), F.A.C. Migration or mixing of fluids from aquifers of substantively different water quality (through the construction or use of a Class V well) shall be prevented by preserving the integrity of confining beds between these aquifers through cementing or other equally protective method acceptable to the Department.

62-528.610 Operation Requirements for Class V Wells:

(1) All Class V wells shall be used or operated in such a manner that they do not present a hazard to an underground source of drinking water.

62-528.630 General Permitting Requirements for Class V Wells:

(4) If at any time the Department learns that an existing Class V well may cause a violation of primary drinking water standards under Chapter 62-550, F.A.C., the Department shall, as determined by following the process in Rule 62-528.100(2), F.A.C.:

(a) Require a permit for such Class V well;
(b) Order the injector to take such actions needed to prevent the violation, including, when necessary, closure of the injection well;
(c) Require monitoring to demonstrate that the water quality criteria in Rule 62-520.420, F.A.C., are not violated; or
(d) Take enforcement action.

(5) Whenever the Department learns that a Class V well may be otherwise adversely affecting the health of persons, the Department shall prescribe action necessary to prevent the adverse effect, including any action authorized under subsection (4). The process for determining these actions is described in Rule 62-528.100(2), F.A.C.
(6) Notwithstanding any other provision of this Chapter, the Department shall take immediate action upon receipt of information that a contaminant which is present or is likely to enter a public water system may present an imminent and substantial endangerment to the health of persons.

because federal regulation exists for injection projects but not for some other recharge methods that also may pose environmental or public health risks—notably, basin recharge by surface spreading and percolation.[19] If EPA were to regulate basin recharge through a program other than the UIC program, some would object to the expansion of the agency's role.

Several conflicting perspectives have been expressed on the issue of the proper federal regulatory role. On the one hand, federal regulation can provide a floor for state programs and may help to prevent pollution that may become a national responsibility (e.g., future Superfund sites). Also, federal resources for research and the development of water quality standards are greater than those of nearly all states. On the other hand, the negative effects of a poorly designed or implemented project are relatively local, there is little competition among states for commercial operators of these projects since they are in response to water demands, and there are other means of funding and disseminating research. Furthermore, in some parts of the country there are states that have more experience and expertise with MUS projects than the EPA does. EPA regulation of MUS remains a relatively small component of a UIC regulatory system dealing with much more significant projects of a different type.[20]

Several additional policy questions arise out of state groundwater protection programs and the application of UIC regulations at the federal and state level, with various regulatory approaches and site-specific issues, particularly where secondary drinking water standards are concerned. Two examples are point-of-compliance and antidegradation policies.

The question of point of compliance arises because regulated constituents may be absent from the injected water at the point of injection, but the water

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[19] As discussed in Chapter 4, infiltration through the vadose zone provides a degree of soil treatment, so recharge basins and vadose zone wells do not present exactly the same risks to aquifer water quality as direct injection. Nevertheless, some contaminants may survive the infiltration process, so it is not the case that injection presents risks while infiltration methods are risk-free.

[20] Efforts at federal and state levels to regulate underground water storage projects encounter the challenge of reconciling the highly variable, site-specific nature of such projects with the need to develop fairly uniform statewide or nationwide rules. To some extent this tension is inherent in the making of laws and regulations, and cannot be relieved completely. As an alternative approach, Seerley (2003, p. 70) recommends building a regulatory regime around the need for extensive site-specific studies:

> "Conclusive data are needed to show how injection/withdrawal schemes, including the consequences of mixing waters of different chemical makeup, may impact hydrogeologic structures as well as the natural systems that depend on groundwater to maintain their long-term biological integrity…. Many states have no regulations in place to require site-specific hydrogeologic studies prior to project implementation, and even fewer address concerns of long-term geologic integrity. Although some of these issues may be addressed in the permitting processes, the statutory language leaves a great deal of room for trial and error rather than creating the structure for a systematic approach that ensures long-term success."

quality can change during aquifer storage. At one location in Florida, for example, injection of water into an underground formation bearing arsenopyrite resulted in arsenic mobilization in the vicinity of the well. To prevent human consumption of water exceeding arsenic maximum contaminant loads (MCLs), and meet the non-endangerment[21] requirements of both the UIC regulations and the Safe Drinking Water Act, the Florida UIC program has requested the EPA to review their proposed arsenic regulations for MUS systems using well recharge (referred to as ASR in Florida's regulations).[22]

Some states prohibit any degradation in water quality in an aquifer, even when both the source water and the water in the aquifer meet all drinking water standards. Such a stringent rule can impede an MUS project by imposing costly pretreatment requirements, or even prohibit MUS altogether. The net benefit to the environment or public health may be very low in comparison to the cost. The California State Water Resources Control Board (SWRCB) proposed a resolution to address the application of antidegradation regulations. SWRCB Resolution No. 68-16 provides for the maintenance of the highest quality of ambient waters and states that any changes to this quality should be consistent with maximum benefit to the people. Box 5-3 presents some questions to consider in balancing antidegradation goals with other benefits, according to SWRCB Resolution No. 68-16.

Strict application of antidegradation policies can raise other risks. Chlorine may be added to water being stored in an MUS project, for example, to ensure compliance with state disinfection requirements for recycled water that may be used for drinking water. The resulting chlorine disinfection by-products (DBPs) may degrade groundwater quality and present a health risk. As the search for additional water supplies becomes more intense, states will be asked to balance the risks posed by their antidegradation policies against the risks posed by alternative water supplies.

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[21] Non-endangerment means that injection operations must not allow fluid containing any contaminants to move into U.S. drinking waters where the presence of the contaminants may cause violations of primary drinking water regulations or adversely affect public health (40 CFR 144.12, as cited in EPA's *State Implementation Guidance—Revisions to the UIC Regulations for Class V Injection Wells*, p. 8)

[22] "Arsenic levels may exceed 10 μg/L within the ASR storage zone under the following conditions: 1. A concentration of 10 μg/L is not exceeded at the property boundary; or 2. The ASR storage zone over the entire area of review contains a TDS concentration of 10,000 mg/L or more; or 3. The ASR storage zone over the entire area of review contains a TDS concentration of 3,000 mg/L or more and a Professional Engineer certifies that the treatment necessary to render the natural water potable will also reduce the arsenic level to 10 μg/L or less; or 4. Institutional controls are in place that prohibit the construction of new drinking water wells used to withdraw water from the ASR storage zone, and there are no existing wells used to withdraw drinking water from the ASR storage zone within the area of review; and 5. Any recovered water is retreated or blended as necessary to meet the water quality standards applicable to the intended use of the recovered water."

BOX 5-3
Balancing Antidegradation and Maximizing Benefits

In California, one criterion is that the lowering of water quality must be to the "maximum benefit of the people of the State." The demonstration of maximum benefit to the people of the state should be considered a balancing test—the greater the decline in water quality, the greater is the required demonstration of benefit to the people of the state. In general, the negative effects of lower water quality must be weighed against the project benefits to assess the net impact on public interests. This is not, however, a formal cost-benefit analysis.

When evaluating the maximum benefit to the people of the state, the benefit should be compared to the alternative of not approving the project. For example, if a water recycling project is not approved, the alternative may be to discharge the treated water to the ocean. Consequently, freshwater supply would have to be used for irrigation instead of recycled water. There would be a monetary cost for using freshwater instead of recycled water. In addition, there would be an environmental cost to develop the freshwater supply, such as the construction of storage facilities or increasing diversion of freshwater supplies from the Sacramento-San Joaquin Delta and other surface waters where beneficial uses are impaired due to diversion-related reduced flows. In some cases, the additional water supply provided by a water recycling project will outweigh the degradation of the groundwater supply. However, this might not be the case if the degradation would impair beneficial uses or had significant impacts on downstream users. In general, a finding of maximum benefit should be difficult when a project shifts significant impacts from one area to another, such as from one portion of a watershed to another.

The analysis of water quality impacts can be complex and should be addressed in environmental impact reports and other environmental analyses. For example, a proposed subdivision that would use recycled water because fresh water is not available may have impacts on groundwater associated with the recycled water use and may have other water quality impacts on surface water associated with urbanization.

Questions to consider when evaluating whether a project provides a maximum benefit to the people of the state include the following:

1. Does the project provide a net environmental benefit? Although the project may cause some lowering of groundwater quality, it may provide offsetting environmental benefits. These may include providing habitat restoration, creating new environmental habitat, avoiding diversion of potable water, preventing seawater intrusion, or augmenting groundwater supplies.

2. Does the project increase the freshwater supply? Projects that replace freshwater use with recycled water use, such as the replacement of freshwater with recycled water for the irrigation of a golf course, augment the freshwater supply, which is a benefit.

3. Does the project prevent the depletion of freshwater supply? Recycled water may be used to supply new water demands, such as irrigation at new parks or residential communities, which would otherwise use freshwater.

4. Would water be used if no recycled water were available? Sometimes water recycling projects are proposed to irrigate sites that would not otherwise be developed. For example, the project may irrigate a new agricultural site to grow an unprofitable crop. These projects should be considered disposal projects and evaluated as such with other disposal alternatives.

5. Is water recycling being proposed as an alternative to providing best practical treatment and control? Sometimes water recycling is proposed as an alternative to providing advanced treatment and discharging to a stream, where the water could also be used beneficially.

SOURCE: California State Water Resources Control Board Resolution No. 68-16 and later supporting documents.

Indirect Potable Reuse via Recharge

Additional or separate regulation programs often are involved for MUS systems that involve reuse of water. There are no federal regulations directly governing water reuse, although state approaches to reuse were summarized in EPA's *Guidelines for Water Reuse* (EPA, 2004). As stated in this document, "As of November 2002, 25 states had adopted regulations regarding the reuse of reclaimed water, 16 states had guidelines or design standards, and 9 states had no regulations or guidelines. In states with no specific regulations or guidelines on water reclamation and reuse, programs may still be permitted on a case-by-case basis" (p.148). In some states (California and Florida), specific programs govern reuse of reclaimed water in MUS systems. Where MUS systems use surface infiltration, the systems are typically treated as indirect potable reuse. Examples of state approaches to indirect potable reuse, particularly for MUS systems, are provided below.

In some states, regulations addressing indirect potable reuse via groundwater recharge are independent from the state's water reuse regulations. For example, in Arizona, the use of reclaimed water for groundwater recharge is regulated under statutes and administrative rules administered by the Arizona Department of Environmental Quality (ADEQ) and the Arizona Department of Water Resources (ADWR). Several different permits are required by these agencies prior to implementation of a groundwater recharge project. In general, ADEQ regulates groundwater quality and ADWR manages groundwater supply. All aquifers in Arizona currently are classified for drinking water protected use, and the state has adopted National Primary Drinking Water Maximum Contaminant Levels (MCLs) as aquifer water quality standards. These standards apply to all groundwater in saturated formations that yield more than 20 L/d (5 gallons per day) of water. Any groundwater recharge project involving injection of reclaimed water into an aquifer is required to demonstrate compliance with aquifer water quality standards at the point of injection.

Sample Water Recycling Criteria Number 1: The State of California

The existing California *Water Recycling Criteria* (California Department of Health Service, 2000) include general requirements for groundwater recharge of domestic water supply aquifers by surface spreading. The regulations state that reclaimed water used for groundwater recharge of domestic water supply aquifers by surface spreading "shall be at all times of a quality that fully protects public health" and that DHS decisions "will be based on all relevant aspects of each project, including the following factors: treatment provided; effluent quality and quantity; spreading area operations; soil characteristics; hydrogeology; residence time; and distance to withdrawal." Until more definitive criteria are adopted, proposals to recharge groundwater using either surface spreading or wells will be evaluated on a case-by-case basis, although draft groundwater recharge criteria described below will guide DHS decisions. California has prepared draft criteria for groundwater recharge (most recently in 2004). The information presented below is based on the most recent draft of the proposed groundwater recharge regulations (California Department of Health Services, 2004); it is likely that substantial changes will be made prior to adoption of the criteria.

While aspects of its regulatory development process have been protracted, California has developed a comprehensive approach to groundwater recharge with reclaimed wastewater. Currently proposed regulations have gone through several iterations and, when finalized and subsequently adopted, will be included in the *Water Recycling Criteria*. The proposed regulations address both surface spreading and injection projects and are focused on potable reuse of the recovered water. The draft regulations, portions of which are summarized in Table 5-1, include requirements for—among other things—source control, water quality, treatment processes, recharge methods, dilution, operational controls, distance to withdrawal, time underground, monitoring wells, and preparation of an engineering report. The criteria are intended to apply only to planned groundwater recharge projects using recycled water (i.e., any water reclamation project planned and operated for the purpose of recharging a groundwater basin designated for use as a domestic drinking water source). They do not apply to wastewater disposal projects.

Constituent Monitoring. The reclaimed water must comply with the following state drinking water regulations: primary maximum contaminant levels, inorganic chemicals (except nitrogen), MCLs for disinfection by-products, and action levels for lead and copper. Quarterly monitoring is required, with compliance determined from a running average of the last four samples. The reclaimed water also must be monitored annually for several secondary MCLs. In addition, the reclaimed water must be sampled quarterly for unregulated chemicals, priority toxic pollutants, and chemicals with state notification levels that DHS specifies based on a review of the project. Each year, the reclaimed water must be monitored for endocrine disruptors and pharmaceuticals specified by DHS after reviewing the project.

Total Organic Carbon. The proposed regulations specify total organic carbon (TOC) as a surrogate for determining organics removal efficiency. Although TOC is not a measure of specific organic compounds, it is considered a suitable measure of the gross organic content of reclaimed water for the purpose of determining organics removal efficiency. The proposed TOC limit is based on increasing concern over unregulated chemical contaminants and the realization that current technology using membranes can readily reduce TOC to 0.2 mg/L or less. The TOC limit applies to TOC of wastewater origin in recharged water. Weekly sampling is required for TOC, with compliance determined monthly from the average of the most recent 20 TOC samples.

Table 5-1 California Draft Groundwater Recharge Regulations

Contaminant Type	**Type of Recharge**	
	Surface Spreading	**Subsurface Injection**
Pathogenic Microorganisms		
Filtration	≤ 2 NTU	
Disinfection	5-log virus inactivation,[a] ≤2.2 total coliforms per 100 mL	
Retention time underground	6 months	12 months
Horizontal separation[b]	150 m (500 ft)	600 m (2000 ft)
Regulated Contaminants		
Drinking water standards	Meet all drinking water MCLs (except nitrogen) and new federal and state regulations as they are adopted	
Total nitrogen	▪ Level specified by DHS for existing project with no RWC increase ▪ ≤5 mg/L for new project or increased RWC at existing project ▪ Or NO_2 and NO_3 consistently met in mound (blending allowed)	
Unregulated Contaminants		
TOC in filtered wastewater	TOC ≤ 16 mg/L in any portion of the filtered wastewater not subjected to RO treatment	
TOC in recycled water	RO treatment as needed to achieve: ▪ TOC level specified by DHS for existing project with no RWC increase ▪ TOC ≤ (0.5 mg/L)/RWC (new project or increased RWC at existing project) ▪ Compliance point is in recycled water or mound[c] (no blending)	100% RO treatment to achieve: ▪ TOC level specified by DHS for existing project with no RWC increase ▪ TOC ≤ (0.5 mg/L)/RWC (new project or increased RWC at existing project)
Recycled water contribution (RWC)	≤ 50% subject to above requirements 50-100% subject to additional requirements	

NOTE: RO = reverse osmosis; RCW = recycled water contribution.
[a]The virus log reduction requirement may be met by a combination of removal and inactivation.
[b]May be reduced upon demonstration via tracer testing that the required detention time will be met at the proposed alternative distance.
[c]If mound monitoring approved.
SOURCE: Adapted from California Department of Health Services (2004).

Dilution Requirements. The draft criteria require a minimum of 50 percent dilution with water of nonsewage origin, although recharge greater than 50 percent reclaimed water may be considered by DHS if certain conditions are met, such as annual testing for tentatively identified compounds (TICs); inclusion of an advanced oxidation process (i.e., hydrogen peroxide addition and ultraviolet radiation); and submission of a proposal and report that includes documentation of compliance with all pertinent criteria, the results of any additional studies requested by DHS, and peer review by an independent advisory panel. The reclaimed water contribution must be determined monthly with compliance based on a running five-year average.

Groundwater Monitoring. Groundwater monitoring wells must be located within one and three months' hydraulic travel time from the recharge area to the nearest downgradient domestic public or private water supply well and at additional points. The monitoring wells must be capable of obtaining independent samples from each aquifer that potentially conveys the recharged water. Monitoring wells must be sampled quarterly for TOC, total nitrogen, total coliforms, secondary MCLs, and other constituents specified by DHS that are identified through reclaimed water monitoring.

Required Permits. Any intentional augmentation of drinking water sources with reclaimed water in California requires two state permits. A waste discharge or water recycling permit is required from a Regional Water Quality Control Board (RWQCB), which has the authority to impose more restrictive requirements than those recommended by DHS, and a public drinking water system using an affected source is required to obtain an amended water supply permit from DHS to address changes to the source water.

The State of Florida

Florida's water reuse rules pertaining to groundwater recharge are summarized in Table 5-2. The rules address rapid-rate infiltration basin systems and absorption field systems, both of which may result in groundwater recharge. Although not specifically designated as indirect potable reuse systems, groundwater recharge projects located over potable aquifers could function as indirect potable reuse systems. If more than 50 percent of the wastewater applied to the systems is collected after percolation, the systems are considered to be effluent disposal systems, not beneficial reuse. Loading to these systems is limited to 230 mm d^{-}(9 inches per day). For systems having higher loading rates or a more direct connection to an aquifer than normally encountered, reclaimed water must receive secondary treatment, filtration, and disinfection and must meet primary and secondary drinking water standards.

TABLE 5-2 Florida Water Reuse Rules for Groundwater Recharge

Type of Use	Water Quality Limits	Treatment Required
Groundwater recharge via rapid infiltration basins (RIBs)	▪ 200 fecal coliforms/100 mL ▪ 20 mg/L $CBOD_5$ ▪ 20 mg/L TSS ▪ 12 mg/L NO_3 (as N)	▪ Secondary ▪ Disinfection
Groundwater recharge via RIBs in unfavorable conditions	▪ No detectable fecal coliforms/100 mL ▪ 20 mg/L$CBOD_5$ ▪ 5.0 mg/L TSS ▪ Primary[a] and secondary drinking water standards ▪ 10 mg/L total N	▪ Secondary ▪ Filtration ▪ Disinfection
Groundwater recharge or injection to groundwaters having TDS < 3,000 mg/L	▪ No detectable total coliforms/100 mL ▪ 20 mg/L$CBOD_5$ ▪ 5.0 mg/L TSS ▪ 3.0 mg/L TOC ▪ 0.2 mg/L TOX ▪ 10 mg/L total N ▪ Primary[a] and secondary drinking water standards	▪ Secondary ▪ Filtration ▪ Disinfection ▪ Multiple barriers for control of pathogens and organics ▪ Pilot testing required
Groundwater recharge or injection to groundwaters having TDS 3,000-10,000 mg/L	▪ No detectable total coliforms/100 mL ▪ 20 mg/L $CBOD_5$ ▪ 5.0 mg/L TSS ▪ 10 mg/L total N ▪ Primary drinking water standards[a]	▪ Secondary ▪ Filtration ▪ Disinfection

NOTE: $CBOD_5$ = carbonaceous biochemical oxygen demand ; TOX = total organic halogen; TSS = total suspended solids.
[a] Except for asbestos.
SOURCE: Adapted from Florida Department of Environmental Protection (1999).

Florida regulations include requirements for planned indirect potable reuse by injection into water supply aquifers. A minimum horizontal separation distance of 150 m (500 feet) is required between reclaimed water injection wells and potable water supply wells. The injection regulations pertain to G-I, G-II, and F-I groundwaters, all of which are classified as potable aquifers. Reclaimed water must meet G-II groundwater standards prior to injection. G-II groundwater standards are, for the most part, primary and secondary drinking water standards. Pilot testing is required prior to implementation of injection projects. Groundwater recharge projects in Florida that involve injection also must comply with the state's Underground Injection Control regulations (Florida Department of Environmental Regulation, 2002), which include criteria pertaining to ASR wells.

Travel Time or Residence Time Criteria

Bank filtration systems in Europe have a precedent for regulation that includes travel time criteria to control pathogens. For example, a "rule of thumb" used in many European countries is that 50 days are typically sufficient to attain water free of pathogens (Grischek et al., 2002). If bank filtration systems exceed these criteria, no disinfection is required of the product water that will be used for potable purposes.

Following similar logic, residence time criteria are being developed at the state level for MUS with reclaimed water. For example, California's proposed groundwater recharge regulations (Table 5-1) specify a minimum residence time that water must be stored underground. The criterion determined for injected water (one year) is longer than that for surface spreading (six months), presumably to address uncertainties in water movement. The required residences times specified in the draft California recharge criteria are based strictly on a review of typical pathogen (specifically virus) inactivation rates and do not consider either site-specific conditions or chemical constituent behavior. A residence time requirement (two years) has also been imposed on an MUS system in Texas with the goal of ensuring virus inactivation in recovered water.

A required residence time prior to withdrawal has the operational benefit of providing a time window for corrective action to be taken in the event monitoring indicates that the reclaimed water does not meet appropriate standards for its proposed use (e.g., potable reuse). Residence times of months have also been shown to be sufficient to attenuate many organic contaminants in groundwater, so such requirements may also be beneficial in this regard, even if this was not the original intent of the regulation.

A limitation of the required residence time approach is its relative arbitrariness with respect to the known important variables among aquifers. Site variables, such as type of aquifer geology and geochemical conditions, significantly impact chemical and microbial contaminant persistence (as described in Chapter 4). Furthermore, in aquifers with flow patterns that are more complex than a relatively homogeneous sand (such as highly heterogeneous or dual-porosity media, for two extreme examples), the high variance of travel times between locations (or residence times in an ASR scenario) may not provide a level of protection comparable to that afforded by flow through a more homogeneous sand system. It is, therefore, not sound science to propose a fixed residence time independent from consideration of site conditions. Currently this is an area of considerable research activity and need.

While rigorous site-specific testing of virus attenuation is not feasible at the field scale at all sites, characterization and consideration of the primary geochemical and microbial characteristics that affect contaminant attenuation are achievable. An alternative model to the required residence time is an "attenuation zone," described in the South Australia Environment Protection Authority's 2004 ASR Code of Practice. Water quality objectives do not need to be met within the defined attenuation zone but would apply outside the attenuation zone. This approach requires that the attenuation be sustainable and, thus, requires a monitoring strategy that demonstrates consistent attainment of treatment objectives.

Interorganizational Coordination

Typically, MUS projects and their regulation require coordination among several organizations, public and private. Projects often involve facilities (treatment plants, recharge and recovery facilities, distribution systems) that are owned, operated, regulated, or otherwise affected by separate public or private organizations, each of which is governed by rules (laws, regulations, charter provisions) specifying what it may, may not, and must do.[23] Extensive monitoring is required—of water conditions above and below ground, consumptive use requirements, species and habitat conditions, and so forth—and those monitoring responsibilities are unlikely to be performed by only one entity (at least in the United States). Furthermore, an MUS project's possible impacts—on overlying lands, hydrologically interconnected surface water bodies, related habitat, water quality, and water use—stretch across the agendas of multiple state and federal agencies.

Involvement of multiple public and private organizations in an MUS project necessitates interorganizational coordination, which includes intergovernmental coordination. "To the extent that policies of one jurisdiction have spillovers (i.e., negative or positive externalities) for other jurisdictions, so coordination is necessary to avoid socially perverse outcomes"(Hooghe and Marks, 2003, p. 239) Interorganizational coordination entails transaction costs—the time, effort, and other expenditures of resources involved in reaching and implementing agreed courses of action.[24] Transaction costs include negotiation and bargaining costs, communication and monitoring costs, and the costs of maintaining and enforcing an agreement.

In an MUS project, because of the involvement of multiple organizations with differing interests and responsibilities, these transaction costs can be considerable. All other things being equal, transaction costs will rise with the number of organizations (public or private) whose actions must be coordinated (Scharpf, 1997, p. 70). Even where the number of organizations is small, disputes over political leadership and authority, the sharing of financial costs associated with a project, and the interpretation of laws and regulations governing

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[23] Even if a single organization were responsible for implementing an MUS project, it would likely have to arrange to use the distribution facilities of a surface water project to deliver water to or from the project site, acquire or perhaps lease land from another organization for the site of the project, and sell the stored water to other clients.

[24] For en overview of transaction costs and their role in institutional analysis, see Eggertsson (1990) and Williamson (1985). For an application of transaction costs in a water management context, see Challen (2000).

different organizations can stall progress toward coordination.[25]

Transaction costs do not exist in a vacuum. Like any cost considerations, they must be viewed in context. There may be offsetting benefits from the involvement of multiple organizations in an MUS project.[26] Public agencies or private organizations that focus on water project operation, water quality monitoring, or administering pumping rights and managing the storage space in an aquifer may exhibit returns to scale or from functional specialization that offset or even exceed the transaction costs of coordination. Organizations such as water associations or special districts at the basin or watershed scale can even ease or overcome coordination problems and enhance the opportunities for MUS.

The existence of transaction costs alone does not argue conclusively for reducing the number of public and private organizations involved in an MUS project; bringing everything under one roof, so to speak, will not necessarily yield overall efficiency gains. What matters is whether the configuration of organizations engaged with an MUS project makes sense in light of the tasks being performed by those organizations, the scale on which they operate, and the constituencies they represent, as well as the transaction costs of coordinating their activities.[27] This is an empirical question that will differ from one location to the next and may change over time in the same location. Accordingly, individuals and organizations contemplating or undertaking an MUS project should be cognizant of the transaction cost implications of interorganizational coordination and be willing to adjust the number, authority, and responsibilities of public and private entities as needed.

In light of transaction costs, a variety of organizational arrangements may support or hinder the practice of MUS. There will not be a single best organizational model for executing an MUS project, and experience within the United States to date indicates that multiple organizational forms are viable: private companies, state agencies, municipal and other public utilities, joint-powers agencies, et cetera. Organizations with overlapping and conflicting interests may or may not overcome their differences in order to move forward with MUS.

One consideration specifically relevant to this issue is the matter of single-

[25] In Monterey County, California, an MUS project to divert and store surplus Carmel River winter flows in the overdrafted Seaside groundwater basin for summer use was delayed for years by a disagreement between the California American Water Company and the Monterey Peninsula Water Management District over control of the project. A demonstration project begun in 1998 had established the feasibility of the operation, but the California Department of Health Services refused to issue a permanent permit for the project until the company and the district arrived at a long-term agreement governing the operation of the project and the disposition of the water (Hennessey, 2005). Such an agreement had not been reached at the end of 2005.

[26] For an application of this idea in the context of metropolitan government in the United States, see Oakerson (1999).

[27] Hooghe and Marks (2003, p. 239) imply the existence of such a trade-off: “The chief benefit of multi-level governance lies in its scale flexibility. Its chief cost lies in the transaction costs of coordinating multiple jurisdictions.”

purpose agencies whose legislative mandates may require them to focus solely on one mission. In some jurisdictions, flood control agencies have such mandates, yet integration of the facilities managed by flood control agencies into an MUS project can be extremely beneficial—even crucial—to its success. The impoundment and controlled release of floodwaters provide some of the greatest opportunities in the United States for moving water to underground storage for later recovery. As the Rosedale-Rio Bravo case described in Box 5-4 illustrates, extreme hydrologic events often provide opportunities for groundwater recharge. Legal or other institutional barriers that inhibit coordination among flood control agencies and other public or private organizations involved in water resource management impose substantial transaction costs with no certain offsetting benefits. In most locations, flood control facilities, surface water storage reservoirs, and underground storage projects can operate in complementary ways or satisfy multiple goals. This is an important matter that warrants further research and institutional reform as needed, in order to exploit the potential complementarities and minimize conflicting operations.

Most communities in the United States are also trying to reduce and control stormwater runoff. As one would expect, a wide variety of agencies and legal instruments are associated with these efforts, from land use regulations promulgated by county or municipal governments to the operation of facilities by special districts. Therefore, integrating stormwater into an MUS program would entail another level of interorganizational coordination.[28] Stormwater quality is extremely variable and often ill-suited for recharging into aquifers: the National Research Council (NRC, 1994) recommended against the use of stormwater from agricultural and industrial areas for groundwater recharge. There may be some possibilities for use of residential stormwater runoff however, and pursuing this potential presents another coordination challenge where the trade-off of transaction costs and potential water management benefits will be faced.

When multiple organizations are involved in an MUS project, as will usually be the case, coordination problems are especially likely to arise over the allocation of benefits and costs among participants. These are not only matters of legal rights to the use or storage of water, which have been discussed above. The allocation of benefits and costs in an MUS project can include which individuals and communities will receive water higher quality or of higher price and for what uses; how the financial costs of constructing, operating, and maintaining facilities associated with an MUS project will be borne; and who shoulders the responsibility for monitoring project operations.

In some cases, municipalities are motivated to develop MUS systems to reduce their dependence on other municipalities for storage and delivery of water. This is particularly true of newer suburban communities that are dependant on large metropolitan centers that have surface water storage facilities and water

[28] For example, stormwater is an important element of groundwater recharge in Southern California cases described in Blomquist (1992).

BOX 5-4
Rosedale-Rio Bravo Water Storage District

The Rosedale-Rio Bravo Water Storage District is located in the southwestern Central Valley of California. When the district was formed in 1959 it contained 43,000 acres. The developed acreage was entirely in irrigated agriculture. The mission of the district was to build and operate a groundwater recharge project to attenuate overdraft, which resulted in water table drawdowns of 8 to 10 feet annually. Historically, recharge was accomplished through the natural percolation of flood flows on the lower Kern River, adjacent to the district. With the completion of flood control projects upstream in the early 1950s, the frequency and magnitude of flood flows diminished, thereby diminishing recharge. Growing agricultural water use also contributed to the overdraft.

Project facilities include approximately 550 acres of spreading grounds along the historic overflow slough of the Kern River. The district succeeded in acquiring supplemental surface water supplies from the federal Central Valley Project, the State Water Project, and the Kern River. The quantities supplied vary significantly from year to year and from season to season depending on runoff conditions. These variable supplies are managed by percolation to the underlying aquifer, which serves as both a reservoir and a distribution system. Surface water deliveries are made to landowners adjacent to project facilities *in lieu* of groundwater pumping and therefore constitute a form of recharge. To date, approximately 2.5 million acre-feet (3 billion m^3) has been recharged since the beginning of the project.

Managed underground storage has reduced water table decline to 2.0 feet annually. Modeling studies show that the water table is 240 feet higher than it might have been without the project. This occurred in spite of a 30 percent increase in water use during the life of the project. Although salts, nitrates, and pesticide residues are present in some areas, the recharging of good quality water has helped to maintain, and in some instances improve, water quality. In recent decades, the district service area has begun to urbanize, and today about 20 percent of the land is devoted to urban and industrial uses. The underlying groundwater is of good enough quality to serve as the basic source of supply for the growing urban uses.

The costs of recharge are estimated to be $79.20 per acre-foot in constant 2004 dollars. The benefits of the project are $1.60 for every dollar of cost. Benefits are attributable to both energy savings and avoided capital costs of additional or deeper wells. It appears that alternative sources of supply are either enormously costly or altogether unavailable. This case study illustrates the importance of groundwater-surface water interactions, and the potential importance of flood flows in recharging groundwater, and it illustrates how MUS can be used to manage highly variable sources of supply and attenuate groundwater overdraft (Roberts and Crossley, 1997).

treatment plants, such as Denver, Albuquerque, Phoenix, Los Angeles, Portland, and Seattle. During periods of drought, these older metropolitan centers may restrict the quantity of water supplied outside their primary service area or may increase their rates. MUS systems can provide smaller communities, particularly those without direct access to surface water supplies, with a means of storing their water locally and obtaining water during wet years, wet seasons, or other non-peak demand periods. Large municipalities may encourage surrounding communities to develop underground storage for conjunctive use of surface and groundwater resources (Hydrosphere Resource Consultants, Inc. et al., 1999).

Nowhere are coordination problems more likely to emerge than around financial issues. Even where MUS appears to be a less expensive alternative water supply development or water storage—that is, even where a "win-win" opportunity appears to exist—the distribution of gains and costs is unlikely to be uniform across participants. Overall benefit-cost calculations may not take adequately into account the difficulties that can arise over questions of who receives the benefits and who pays the costs. This is only one of the institutional issues that touches upon the economic considerations involved with water resource management.

ECONOMIC ISSUES

The Economics of Groundwater Management

The economics of groundwater management and use has been well developed. There is a substantial and varied literature on the economics of groundwater use that develops and characterizes a set of common principles upon which economically efficient management and use can be based (see, for example, Burt, 1970; Cummings, 1970; Gisser, 1983; Burness and Martin, 1988; Provencher and Burt, 1993). Fortunately, the economics of managed underground storage can easily be integrated into the framework that these principles provide. The general economic prescription for efficient groundwater use requires that water be extracted at rates where the net benefits (total benefits net of total costs) are maximized over time. Benefits are determined by the uses to which the water is put. In the short run, costs include the cost of extracting the groundwater and the opportunity cost, which is frequently called a user cost.

Extraction costs depend on the cost of energy, the depth from which the water must be pumped, and the efficiency of the pump. Opportunity costs reflect the cost related to extracting and utilizing the water now compared to conserving it for later use—for example, water pumped in the current period results in a lowered water table for all future periods. If extractions are to be efficient, pumpers must account, through the user cost, for the consequences of extractions on future water table levels. Economically efficient extraction leads to an optimal water table depth when the steady state is reached. An optimal steady-state depth is reached when all pumpers account fully for all of the costs of extraction, including the user cost.

Much of the economic literature on groundwater focuses on the case where the resource is treated as a common pool and extractions tend to occur at rates that are inefficient, with the result that too much is pumped too soon and steady-state water table levels are lower than optimal. When groundwater is treated as a common pool resource, pumpers have an incentive to ignore the user costs; this results in a steady-state depth to the water table that is deeper than optimal.[29]

In an overdrafted aquifer there may be costs in addition to the increased costs of pumping from a lowered water table. These can include the costs of land subsidence and an increased risk of saltwater intrusion in coastal aquifers. There can, however, be circumstances in which overdrafting is economically efficient. This occurs when the benefits of use are quite high in relation to the costs of extraction, which is often the case during severe droughts. In most instances, overdrafting will be economically efficient only on an intermittent basis.

On the other hand, persistent overdraft is always self-terminating. As water tables decline, eventually a point is reached where the costs of additional extractions are greater than the benefits associated with any of the uses to which the water may be put, at which point the aquifer is said to be "exhausted economically" even though it still contains some water. When it is no longer economical to extract water, pumpers either stop extracting it or reduce the quantity extracted. This process continues until extractions equal recharge and thus the quantities extracted are exactly equivalent to the safe or the sustainable yield. In this circumstance, the overdrafted aquifer reaches a steady-state equilibrium. When groundwater is not recharged, as is the case with fossil groundwater, f over time should occur in a pattern that maximizes the present value of the net benefits. Ultimately, a point will be reached where the benefits from extraction are less than the costs of extraction. At this point, the aquifer is said to be, "economically exhausted." Economic exhaustion is quite different from physical exhaustion or complete dewatering. Aquifers are rarely completely dewatered.

When groundwater is exploited in an individualistically competitive fashion, the rates of extraction and the resulting steady-state depth will usually not be optimal. The user costs are typically ignored, because pumpers believe that their own extractions have a very small impact on other pumpers and because they perceive that voluntary restraints on extraction serve only to make the water available to competing pumpers. The problem of optimal regulation in situations where groundwater is exploited competitively is to provide incentives that will cause pumpers to behave in the aggregate in a way that takes user costs into account. Corrective measures that can be employed to help achieve this result include the formal vesting of property rights to groundwater in situ, pumping quotas, and pump taxes that are set equal to the marginal user cost. There are some instances in which the transmissive properties of the aquifer are such that extractions by one pumper will have no impact on adjacent pumpers. In such

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[29] It should be noted, however, that the common pool characteristics of aquifers—and the impacts pumpers have on one another—follow primarily from simplifying assumptions that the modeled aquifer is relatively transmissive and nonsegmented. The effects that pumpers' withdrawals actually have on one another will depend on specific characteristics of an aquifer, and these considerations have been taken into account only occasionally in past economic analyses.

cases, corrective measures are unnecessary.

It is important to note that the formal economics of groundwater management should account for groundwater-surface water interactions. Groundwater extractions can diminish or eliminate discharges to the surface. Thus, for example, Glennon (2002) documents a number of cases in which the failure to consider groundwater-surface water interactions led to serious adverse outcomes. Moreover, Alley and Leake (2004) note that the failure to account for these interactions may lead to erroneous estimates of steady-state or sustainable withdrawals. Strictly speaking, equilibrium is achieved when recharge equals withdrawals, where withdrawals include extractions plus discharges. Inasmuch as MUS entails groundwater storage and is not intended to affect groundwater-surface water interactions, the discussion that follows focuses on the economics of MUS and abstracts from groundwater-surface water interactions.

The formal economics of groundwater management and use reveals an important conclusion for MUS. When groundwater is treated as a common pool resource, the incentive to invest in MUS facilities and operations is eroded. Water recharged and stored is freely available to competing pumpers who need not pay to capture it. Thus, an important lesson for the development of successful underground storage schemes is that the aquifer in question must be managed in ways that prevent it from being treated as a common pool resource.

The economic implications of groundwater quality should not be neglected. In general, aquifers possess significantly less capacity to process waste and self-cleanse than surface waters. This means that once groundwater is contaminated, it remains contaminated for very long periods. There are methods to accomplish groundwater remediation or cleanup. Although in situ methods show some promise, the conventional remediation technique entails pumping and treating. The costs of pumping and treating or any groundwater cleanup regime are very high. They are so high, in fact, that the economics of groundwater quality can be resolved into the simple proposition that it is almost always cheaper to prevent the contamination of groundwater in the first place than it is to clean up once it has occurred. This principle will almost always hold for MUS projects given that the project requires investment that may be considerable. Preventing contamination of the aquifer that is utilized for storage is a matter of protecting that investment as well as avoiding the high costs of cleanup.

The Economics of Managed Underground Storage

The operation of underground storage schemes will usually require the use of artificial recharge operations. The economics of artificial recharge for direct use has been analyzed in detail by Brown and Deacon (1972), Cummings (1971), and Vaux (1985). Briefly, artificial recharge augments the rate of recharge and thereby increases the quantities of water that can optimally be extracted in a given period. Artificial recharge can also reduce uncertainty about supplies, arising from variability in surface water flows. The economic justification for artificial recharge is that its costs may be less than the benefits that accrue from the various uses to which the recharged water can be put. Artificial recharge also may reduce the vulnerability of an aquifer to saltwater intrusion or to subsidence, and the economics of these considerations have also been addressed in a number of case studies (Cummings, 1971; Warren et al., 1975).

Traditional means of coping with water scarcity and hydrologic variability have been to construct surface water reservoirs that allow water to be captured and stored in wet times and places so that it can be made available for use in dry places during dry times. Today, our ability to construct additional surface storage capacity is sharply constrained by reduced land availability, rising construction costs, and ecological impacts (see Chapter 1). Yet population and economic growth have led to intensifying water scarcity, and additional storage would help to alleviate that. There are a number of pioneering examples of the use of MUS in lieu of surface water storage or to help manage highly variable flows from surface water sources, including storage. Two of these in California, the Arvin-Edison Water Storage District and the Rosedale-Rio Bravo Water Storage District, are described in Boxes 5-4 and 5-5, respectively.

Historically, storage space in aquifers has not been treated as if it were a scarce commodity. Rather, in the face of the very large capital costs of surface water impoundment facilities, both public and private operators have sought long-term water supply contracts that significantly reduce the probabilities of a financial default. Typically, markets for storage capacity are thin or nonexistent. One result is that there is a dearth of empirical data on the scarcity value and economics of storage capacity.

In regions where water is scarce and the scarcity is intensifying, it is reasonable to assume that the value of storage capacity is significant. It is estimated, for example, that at the margin, storage capacity in California has a value of about \$600-\$800 per acre-foot per year (Richard E. Howitt and Jay R. Lund, University of California, personal communication with H. Vaux, 2006). It seems likely that values that high would be found throughout the arid and semi-arid West and in other regions where water is locally constrained and quite scarce.

Finally, it is important to draw the distinction between the economics of groundwater management and the financing of groundwater management. The economics of groundwater management is about the full range of costs and benefits and the values that attach to those costs and benefits. Finance is about the monetary or pecuniary aspects of groundwater management— such issues as (1) where investment capital is to be obtained and at what cost, and (2) how capital is to be retired and the cost of capital repaid. Economics focuses more broadly and transcends financial issues. It is possible for a project or program to be financially justified but not economically justified. This might be true if there were large external costs, such as environmental damages, that did not have to be compensated monetarily. Conversely, it is possible for a project to be economically justified but not financially justified as, for example, when problems of risk and cash flow in the early stages of the project make it impossible to

BOX 5-5
The Arvin-Edison Water Storage District:
Managed Underground Storage for Agriculture

The Arvin-Edison Water Storage District occupies approximately 132,000 acres (53,000 hectares) in the extreme southeasterly portion of California's Central Valley. While soils are rich and the growing season is favorable, average annual precipitation is just 8 inches (200 mm) and inadequate to support rain-fed agriculture. Thus, successful agriculture depends almost wholly on the availability of water for irrigation. Early growers in the region irrigated exclusively with groundwater.

With favorable growing conditions, irrigated acreage expanded and overdraft became severe and persistent. By the 1930s, overdraft amounted to 113,000 acre-feet (140 million m^3) annually. It became clear that supplemental sources of surface water would have to be found if agriculture was to continue on its existing scale. The Arvin-Edison Water Storage District subsequently contracted with the federal government to supply water imported from the north via the Central Valley Project. The contract entitled the district to 40,000 acre feet (49 million m^3) of annual firm supply and 311,675 acre-feet (384 million m^3) of interruptible supply. Thus, only 11percent of the supplemental supply was reasonably reliable, with the remaining 89 percent delivered on an "as available" basis that depended on higher than average levels of precipitation. At this point, the district's need was not so much for additional water as it was for more reliable water.

Arvin-Edison was able to acquire some additional firm water through a series of surface water exchanges, but these were insufficient to resolve the problem completely. The district then developed a conjunctive use program, which allowed it to store underground excess (non-firm) supplies in wet years and utilize them in dry years. Between 1966 and 1999 the district stored a total of 4.2 million acre-feet (5.2 billion m^3) in the underlying aquifers. This storage had a number of benefits. First, groundwater levels have been stabilized through a combination of reduced extractions and a formal program of recharge. This means that the costs to those who continue to extract groundwater are less than they would have been in the absence of the formal recharge program. Second, the surface water service area accounts for only about 40 percent of district lands. Growers on the remaining 60 percent do not have access to surface water and continue to extract groundwater. In effect, the stabilization of groundwater levels permitted the district to continue to serve a significant proportion of its users with groundwater and thereby avoid the considerable expense of a surface water delivery system for the groundwater service area. Third, in the area that is served with surface water, groundwater is available to growers in drought years when surface water deliveries are reduced. The groundwater is pumped from wells maintained by the district and introduced into the surface water delivery system.

A simple cost analysis shows that stabilization of groundwater levels has resulted in substantial cost savings. By assuming that (1) groundwater pumps in the District have an average efficiency and (2) the marginal cost of energy per kilowatt-hour is between $0.12 and $0.20, the savings to pumpers from not having to lift water the additional 235 feet (80 m), which would be the case had there been no recharge program, range between $47 and $80 per acre-foot. Total water costs to Arvin-Edison users in 2000 amounted to $79.90 per acre-foot. Water costs, then, are between one-half and two-thirds of what they would have been had groundwater levels not been stabilized, and all growers continued to extract the same quantities that they had extracted historically in spite of the increased costs.

This example illustrates how agricultural water users can benefit from sustainable underground storage. The fact that the aquifer in question is hydrologically isolated proved to be an important pre-condition. Because of the hydrologic isolation, there were no competing pumpers who would have been in a position to reap the benefits of the recharge project as free riders. The hydrologic circumstances of the aquifer effectively restricted those who could benefit from the recharge to those who paid for it. Thus, it was not necessary to adjudicate groundwater rights prior to undertaking the recharge project. This saved significant time and much money.

attract the funds needed for completion and the transition to full operating status. The next section focuses on the economics of groundwater management. The financing of groundwater management and managed underground storage is discussed fully in Chapter 6. That discussion identifies the critical variables affecting financial feasibility and generally characterizes the importance of financial drivers in determining the feasibility of specific managed underground storage projects

The Economics of Multiple Objectives

There are several possible objectives for any project or process of artificial groundwater recharge. First, such recharge is frequently done for the purpose of augmenting the quantity of water in storage. This objective has become increasingly attractive as the opportunities for surface water storage have diminished and the environmental and other costs of surface water storage projects have risen. Second, artificial recharge may also be undertaken in an effort to stabilize groundwater levels. Thus, for example, where water tables decline continuously because an aquifer is overdrafted, artificial recharge is one means of augmenting total recharge and either bringing extractions into balance with recharge or narrowing the difference between the two. Third, artificial recharge may be used to mitigate or avert some of the costs of persistent overdraft (e.g., land subsidence, seawater intrusion). Fourth, artificial recharge can be used to control the migration of contaminant plumes, thereby protecting the quality of the groundwater. These objectives tend to be interrelated: that is, measures focused on the achievement of one of the objectives often result in the achievement of one or more of the others.

This does not mean that all effects of artificial recharge are beneficial. For example, artificial recharge for the purpose of augmenting storage could lead to flooding of basements and other subterranean structures in very wet years or raise water tables to a level where contaminants are mobilized from soil layers near the land surface. In planning for artificial recharge it is important to acknowledge explicitly the possibilities for achieving multiple objectives, as well as to account for potential adverse impacts. Ideally, an artificial recharge program should be planned so that total net benefits, those related to all objectives, are maximized.[30]

[30] There is a substantial literature on the methods of multiobjective planning (e.g., Loucks and van Beek, 2005). It is customary to employ methods that either optimize the mix of emphases on the different objectives or entail achieving a set of targets. Target planning entails the identification of plans that best meet a predetermined mix of objectives or targets. Optimization planning also requires prior knowledge of the decision maker or policy maker's preferences but requires that these preferences be expressed in terms of objectives rather than targets. The goal of optimization planning is to identify the optimal mix of objectives that can be achieved subject to a set of financial and other feasibility constraints.

As a general rule, MUS will require explicit identification and consideration of all objectives and costs, both actual and potential. Underground storage projects are more likely to be sustainable if they are conceived and operated in fashion in which future circumstances have been foreseen and flexibility is maintained to permit adaptation to circumstances that cannot be foreseen. The quality of water in a given aquifer may not be threatened currently by the proximity of a contaminant plume, for example, but such an eventuality could arise in the future and the costs of addressing it may be significantly reduced if the recharge system is adaptable and flexible. It is also true that the presence of multiple objectives may make an underground storage project more economically attractive than if there were only a single objective. The conclusion is that for economic reasons and to promote sustainability, underground storage plans should account for all objectives and their costs and benefits.

Spillovers and Unmarketed Benefits

In modern, highly complex market systems with millions of interrelated actions, market imperfections are common. Such imperfections may introduce significant distortions into observed economic behavior and need to be accounted for in designing water supply or water delivery projects, in the economic analysis of the costs and benefits, and in financing. Two common market imperfections are spillovers—often called "externalities"—and the presence of unmarketed or misvalued benefits. These imperfections are likely to be present with some frequency in MUS projects.

Spillovers or externalities are said to occur when an economic transaction results in impacts on a person or persons who are not party to the transaction. There are both negative externalities, which inflict costs on those not party to the transaction, and positive externalities, which confer benefits. The general conclusions about externalities are quite straightforward. Where external costs are present, the good or service tends to be overproduced or overconsumed relative to what would be economically optimal (e.g., extraction of groundwater by one producer lowers the water table for all others). Where external benefits are present, the good or service tends to be underproduced relative to what would be economically optimal because of the inability of the private investor to capture all of the returns from the investment (e.g., one producer recharging an aquifer when stored water can be extracted by anyone). Usually, therefore, restraint of pumping or provision of recharge will have to be produced through a public

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In the case of target planning the goal is to attain the target values without reference to constraints. Optimization planning acknowledges the existence of constraints of all sorts. In general, formal mathematical methods of multiobjective planning require that objectives and constraints be quantified.

entity or an institution such as a user cooperative that has the authority to regulate users' behavior and/or to tax or otherwise secure payment for the recharge service from all those who benefit.

The general remedies for externalities include taxes (and subsidies) and regulations. In general, taxes are the most straightforward and are set at the marginal value (cost) of the external cost. When the tax is added to the unadjusted price, the externality is appropriately reflected in the price and economically efficient levels of production and consumption occur, other things being equal. In some circumstances, appropriate subsidies can accomplish the same thing, encouraging or compensating one who produces a beneficial externality.

Regulations can be used to accomplish the same outcomes, but in general they are harder to design, may entail significant enforcement costs if they are to be effective, and are difficult to fashion so that they both are effective and accommodate differences in the circumstances of different producers and consumers. In principle, regulations are thought to be superior to pricing incentives only in circumstances where it is not possible to measure the magnitude of the spillover or externality or where the magnitude is so large that catastrophic impacts are a possibility (Baumol and Oates, 1979). In practice, however, regulations are employed more frequently than taxes or price incentives.

When markets function reasonably well and imperfections are absent or minor, prices provide an accurate guide to the value of goods and services that are traded in those markets. For goods and services that are not traded in markets, prices are absent and the value of such goods and services is not immediately obvious. Water itself is rarely priced in markets. The prices paid by most water users reflect the costs of capturing, storing, and conveying the water and of treatment in the case of domestic supplies. In other words, since water is not often traded in markets, it tends to be assigned a scarcity value of zero and is treated as if it were a free good. This signals consumers that water is much more freely available than it is in fact. Consumers do not face prices that reflect the true scarcity value of water. This means that water is used in quantities that exceed the economically efficient quantity.

Other relevant nonmarketed products include environmental services and environmental amenities. Glennon (2002) documents in detail the connection between groundwater and environmental amenities and services, showing that groundwater depletion has significant adverse impacts on the values of these amenities and services. Glennon also notes that the unmarketed nature of environmental amenities and services means that there is a tendency to undervalue them or ignore them altogether. Inasmuch as artificial recharge and augmentation of storage may have positive impacts on environmental amenities and services, it is important to recognize the need to value these and other benefits that may not be traded in markets.

The fact that water itself rarely has a market-determined scarcity value means that comprehensive economic valuation of artificial recharge schemes will require the use of alternative valuation methods. Acceptable valuation methodologies exist and are used to value an entire range of unmarketed goods and services (NRC, 1997, 2005). These methods include inferential techniques in which the value of a good or service can be inferred indirectly from the behavior of consumers and survey techniques that query consumers about their valuation of certain nonmarketed amenities. Economic analyses of MUS proposals will frequently require the use of such methods to value benefits and costs.

Comparative Values and Costs

The costs and values of MUS are necessarily relative. The cost competitiveness of a given project cannot be determined in any absolute sense. The problem is compounded by the fact that storage capacity is rarely priced according to its scarcity value. The financial realities of water project construction and operation mean that storage tends to be allocated through long- term contracts that are executed at the outset and rarely renegotiated when they expire (Bain et al., 1966). This financial practice ensures that the project costs or a portion of them are repaid over the life of the project. While there is financial justification for such practices, they have the effect of shielding storage capacity from the economic forces of competition. This means that storage is underpriced or not priced at all and that the financial costs of storage projects understate the economic costs by a least the scarcity value of the storage.

Scarcity costs aside, the relative attractiveness of any storage project will depend on the costs of other alternatives as well as the value of the use to which the water is to be put. Thus, for example, the costs of MUS at the Orange County Water District are in the range of $400-$600 per acre-foot which in any absolute sense appears relatively high. Yet the cost of the cheapest alternative source of water—imported water purchased from the Metropolitan Water District of Southern California—is on the order of $650 per acre-foot and the costs of other alternatives, such as seawater desalting, are even higher. In the circumstances faced by the Orange County Water District, MUS is attractive from a cost standpoint even though the costs of treating the water to be stored are relatively high.

The relative value of the uses to which the water is put is also important. In the Orange County case, the project is attractive not just because the relative costs are low but because the water is put to domestic, industrial, and commercial uses, all of which are relatively high-valued. As a general rule, these uses are valued higher than agricultural uses and many environmental uses, although some environmental uses appear to have sizable values. The Orange County Project would not look so attractive, for example, if the water was to be used to irrigate fodder crops, a relatively low-valued use. In that circumstance the costs would likely be significantly higher than the value of the use and would raise compelling questions about the economic justification of the project. The result is that the attractiveness of any MUS project depends on the costs of alternative sources of supply as well as the value of the product water in its final uses. Financial considerations are discussed more fully in Chapter 6.

For these reasons, it is difficult to make generalizations about the attractiveness of MUS, since it will depend almost exclusively on local or regional water supply and water use conditions. Nevertheless a few generalizations can be made. Managed underground storage is more likely to be an attractive option when the value of the final use is high. It is likely to be a competitive option where alternative sources of water supply are either unavailable or very costly. It is also likely to be attractive when the costs of treating the original source water to appropriate levels of quality are low. Managed underground storage is likely to be far more attractive in the future because low-cost water supply options are no longer available in many regions and locales and, because high-valued uses are growing in many expanding urban areas and in those regions where source water can be obtained relatively inexpensively and costly treatment can be avoided.

Subsidies

Frequently, the high costs of providing water supplies or remediating and enhancing water quality result in calls for public subsidy in order to make the project or program "affordable." Often, advanced techniques of augmenting water supplies such as desalination, wastewater reuse, or groundwater recharge appear very costly in comparison with the costs of established alternative water sources. The relatively higher cost of "new" water invariably leads to demands for public subsidization in order to keep the costs of all water supplies roughly equivalent. From an economic perspective it is important to understand the circumstances in which subsidies are warranted and those in which they are not.

The general rule is that where the value of goods and services is totally reflected in the price, there is no economic justification for subsidization. Nevertheless subsidies are used for a variety of purposes. Some subsidies are designed to restrain production, keeping the subsidy-adjusted price higher than would be the case if prices were determined by market forces alone. Other types of subsidies lead to prices that are lower than those that would result if market forces were left untouched. In these circumstances, a subsidy simply represents a gift in the form of an artificially low price. Also, there are mechanisms such as average cost[31] pricing that keep the price of utility services—electricity, gas and water—lower than they might be otherwise. When subsidies are used to depress artificially the price of some good, that good will be produced and consumed in quantities that are greater than the economically efficient quantity. The justification for these subsidies invariably rests on social and political, not economic,

[31] Average cost is total cost divided by the number of units of output. It is the average cost of producing each unit of output. The marginal cost is the cost of producing one additional unit of output

grounds. Frequently, for example, subsidies may be required to ensure that a project is financially feasible. As a consequence, where financial feasibility is an overriding concern, subsidies may be common. Subsidies in the context of financial feasibility are discussed further in Chapter 6.

There are certain instances in which subsidies may be justified economically. These are cases where the market-generated price of the good or service does not fully reflect its value. The earlier conclusion that investment in groundwater recharge facilities and operations would be less than optimal if left to the private sector is a case in point. Where groundwater is extracted competitively, all extractors benefit from the recharge in the form of reduced levels to the water table and consequent reduced pumping costs. Yet, a purely private entrepreneur cannot capture all the returns from these benefits and thus invests less in the recharge operation than is optimal. In the absence of some other collective arrangement that would allow all of the returns to be captured by the investor, subsidizing investment in recharge facilities would be one method of securing more nearly optimal levels of investment. Another pertinent example is the case where an artificial recharge operation augments storage and repels the advance of a contaminant plume thereby protecting the quality of the groundwater for all pumpers. In this instance, protecting its quality for one protects the water quality for all, and the gain in water quality protection cannot be withheld from an extractor who refuses to pay for it. In such instances a subsidy to the recharger that reflects the total benefits from recharge would be economically justified. Alternatively all extractors could be taxed for the amount of the benefit. The choice between a public subsidy and an alternative institution would depend in part on which alternative entails the smallest transactions and administrative costs.

The conclusion is that subsidies are justifiable on economic grounds in circumstances where market prices do not capture all of the values—both positive and negative—of some good or service. Where subsidies lack an economic justification, they will distort prices and affect the allocation of goods or services in ways that are less than economically optimal. Such subsidies should be established carefully since in some cases subsidization encourages water use and this may not always be desirable where water is scarce.

CONCLUSIONS AND RECOMMENDATIONS

Conclusion: Some states have created statutory schemes that are tailored to MUS projects; this approach is desirable because of the novel questions posed. For example, a state may find it desirable that withdrawals from an MUS project be done over a longer time period than a traditional water right might provide or that MUS be allowed despite the junior status of the right's holder. States can anticipate these adjustments to traditional water rights as appropriate.

Recommendation: While a comprehensive approach has advantages, at a minimum states should define property rights in water used for recharge, aquifer storage, and withdrawn water, to provide clarity and assurance to MUS projects.

Conclusion: The federal regulatory requirements for MUS are inconsistent with respect to treatment of similar projects. Federal UIC regulation addresses only projects that recharge or dispose of water directly to the subsurface through injection wells, while infiltration projects are regulated by state governments whose regulatory standards may vary. The appropriateness of regulation through the UIC program has been questioned by states with active ASR regulatory programs. Also, there are inconsistencies between the Clean Water Act and the Safe Drinking Water Act that impact MUS systems. For example, some jurisdictions try to control surface water contamination problems by diverting polluted water from aboveground to groundwater systems. This approach may undermine MUS programs by putting contaminants underground without appropriate controls.

Recommendation: The federal and state regulatory programs should be examined with respect to the need for continued federal involvement in regulation, the necessity of a federal baseline for regulation, and the risks presented by inadequate state regulation. A model state code should be drafted that would assist states in developing comprehensive regulatory programs that reflect a scientific approach to risk.

Conclusion: Regulations are, quite properly, being developed at the state level that will require a certain residence time, travel time, or travel distance for recharge water prior to withdrawal for subsequent use. However, regulations based on attenuation of a single constituent or aquifer type, such as pathogen attenuation in a homogeneous sand aquifer, may not be appropriate for a system concerned with trace organics and metals in a fractured limestone, and vice versa. Such regulations are particularly pertinent for MUS with reclaimed water.

Recommendation: Science-based criteria for residence time, travel time, or travel distance regulations for recharge water recovery should be developed. These criteria should consider biological, chemical, and physical characteristics of an MUS system and should incorporate criteria for adequate monitoring. The regulations should allow for the effects of site-specific conditions (e.g., temperature, dissolved oxygen, pH, organic matter, mineralogy) on microbial survival time or inactivation rates and on contaminant attenuation. They should also consider the time needed to detect and respond to any water quality problems that may arise.

Conclusion: MUS projects can exhibit numerous and complementary economic benefits, but they also entail costs. Some of those benefits and costs are unlikely to be incorporated in the calculations of individual water users—that is, there may be spillover costs to third parties or spillover benefits that are not given market valuations. Failure to account for all benefits and costs, including ones that may not be reflected in market prices for water, can lead to underinvestment in groundwater recharge, overconsumption of water supplies, or both.

Recommendation: An economic analysis of an MUS project should capture the multiple benefits and costs of the project. MUS projects invariably entail the achievement of multiple objectives. Third-party impacts, such as the environmental consequences of utilizing source water, should be included.

Conclusion: Water resources development has been characterized by substantial federal and state subsidies. As water shortages intensify, the political pressure for investment in new technologies will increase.

Recommendation: Water managers should avoid the introduction of further distortions in prevailing choices of water technologies. To ensure optimal investment in MUS and other technologies, subsidies should be provided only when there are values that cannot be reflected fully in the price of recovered waters. An example of such a value would be an environmental benefit that accrues to the public at large. In particular, simply lowering costs should not be the justification for providing subsidies for MUS projects.

Conclusion: Antidegradation is often the stated goal of water quality policies, including policies that apply to underground storage of water. For any MUS project – including storage of potable water, stormwater, and recycled water – it is important to understand how water quality differences between native groundwater and the stored water will be viewed by regulators who are charged with satisfying those regulatory mandates. In addition to water quality factors, a broader consideration of benefits, costs, and risks would provide a more desirable regulatory approach. Therefore, weighing water quality considerations together with water supply concerns, conservation, and public health and safety needs is an essential plan of action. Rigid antidegradation policies can impede MUS projects by imposing costly pretreatment requirements and may have the practical effect of prohibiting MUS even in circumstances where the prospects of endangering human or environmental health are remote and the benefits of water supply augmentation are considerable.

Recommendation: State laws and regulations should provide regulatory agencies with discretion to consider weighing the overall benefits of MUS while resolutely protecting groundwater quality.

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6
Project Development, Monitoring and Management

The general components of a managed underground storage (MUS) system and some of the broad decisions driving the selection of the type of system to be developed are addressed in Chapter 2. Hydrogeological and groundwater engineering issues; water quality and geochemical issues; and legal, institutional, and economic issues related to the development, operation, and management of MUS systems are addressed in Chapters 3, 4, and 5, respectively. This chapter provides an opportunity to address some of the issues that face project proponents (and opponents) and managers of MUS systems that have not been discussed earlier.

It should be noted up front that the entire project development, monitoring, and management process is likely to be more successful if a broad approach to water resources planning is taken. As described in Chapter 7, an integrated strategy in which all measures for managing water scarcity are considered carefully in a systematic way is highly recommended. Such a strategy would ideally include metrics to allow comparisons of MUS to other water supply and storage options. It would also take into account watershed-wide water quality management, including control of stormwater, combined sewer overflows, septic tank leaks, agricultural runoff, and coastal water quality issues. The formation of regional water authorities can be a useful step in understanding and incorporating planning in the context of the regional and state water supply and water management options.

MUS is not always the solution—or even part of the solution—to water supply and storage challenges. The methodologies described in this chapter assume that all of the options for a given area have been evaluated thoroughly. The list of the geological, hydrologic, geochemical, geotechnical, environmental, public health, water availability, economic, regulatory, and other issues that need to be considered to evaluate the potential suitability of MUS is long; many of these issues have been described in Chapters 3, 4, and 5. No comprehensive MUS site comparison exists; the closest—but an extensive review of aquifer storage and recovery (ASR) planning methodologies—was performed by Brown (2005).

The chapter is divided into two main parts. The first part is a summary of the major steps—beginning to end—in the selection, development, management, and oversight of MUS systems. The second part provides an expanded discussion of four key issues that a manager, operator, or regulator may need to consider. These issues are clogging, monitoring (including the use of surrogates or indicators), public perception, and financing.

FROM FEASIBILITY TO CLOSURE: STAGES OF AN MUS PROJECT

This is a summary of key steps or stages of project development, rather than a "how-to" manual. Many fine references exist on the practical issues of ASR, surface infiltration, and artificial recharge in general. Pyne (2005) discusses in detail the building blocks of an ASR program, including feasibility studies, pilot testing, well design and equipment, plugging issues, and water quality challenges. Brown (2005) does an excellent job of summarizing existing frameworks for both brackish and freshwater ASR projects. He presents a much more detailed framework than the one used in this chapter, including flow charts for the various steps.

The Environmental and Water Resources Institute (EWRI, 2001) offers another useful and practical guidebook. The Water Environment Federation and the AWWA (1998) provide an overview of planning indirect potable reuse, health and regulatory considerations, treatment technologies, system reliability, and public information outreach programs. Segalen et al. (2005) give a fine summary on that topic.

Several recent scientific meeting proceedings are an excellent source for case studies; notable among these are those of the forth and fifth International Symposia on Management of Aquifer Recharge (Dillon, 2002; UNESCO, 2006) and U.S. Geological Survey (USGS) Artificial Recharge Workshop Proceedings (Aiken and Kuniansky, 2002). Finally, Ruetten (2001) deals in depth with public perception issues.

As noted above, there are many different ways to organize an MUS project from beginning to end. The following list is modified only slightly from EWRI/ASCE (2001) *Standard Guidelines for Artificial Recharge of Ground Water*:

- Phase I: Preliminary activities (also called feasibility evaluation), including data collection; assessment of regulatory, legal, political, and economic feasibility; and conceptual planning; this phase may also involve environmental assessment and public involvement
- Phase II: Field investigations and testing of pilot sites
- Phase III: Design (revision of the conceptual design to reflect results of investigations)
- Phase IV: Construction and start-up (systems may require cycle testing to develop recharge systems)
- Phase V: Operation and maintenance
- Phase VI: Project review and adaptive management
- Phase VII: Closure

Phase I: Feasibility Evaluation

The components typically addressed in a feasibility evaluation for an MUS system are summarized as follows:

- *Site assessment*—land availability, ownership, proximity to water sources, suitability of aquifers, preexisting groundwater quality, proximity to water use;
- *Legal and regulatory issues*—water rights for source water and ownership of water in storage, antidegradation requirements, monitoring requirements;
- *Financial considerations*—cost of land, treatment and conveyance facilities, access to capital, availability of grants, loans or other subsidies, costs of operation, and ability to obtain revenue from water users;
- *Purpose of MUS* (duration expected) —seasonal or long-term storage, drought protection, meet summer peak demands, or evening out surface water treatment plant flows;
- *Source water availability and quality*—Raw surface water, stormwater, recycled water or treated drinking water, high suspended solids load or clear water, quality comparison to existing groundwater; total dissolved solids (TDS), pH, redox potential, trace elements, microbial quality, trace organic contaminants;
- *Treatment needs and existing capacity*—desilting to prevent clogging by suspended or settleable solids, pH adjustments or other methods to reduce adverse changes in metal concentrations, reverse osmosis for removal of salts and trace organics, and advanced oxidation for destruction of trace organics; unused capacity of existing treatment facilities;
- *Capture and conveyance facilities needs*—stormwater capture impoundments, temporary storage prior to recharge, pipelines or channels for conveyance, pumping needs; and
- *Public perception*—outreach program needs, existing perceptions of groundwater quality, different outreach needs depending on source water, especially reclaimed water, different outreach depending on perceptions of water resource needs and impacts of the MUS project, (e.g., drought protection vs. growth inducement). As noted later in this chapter, this step is sometimes left for later in the planning process, to the detriment of all concerned parties.

The feasibility of recharging water into the aquifer is perhaps the key issue to explore in a feasibility evaluation. If water cannot be recharged in sufficient quantities, the project will not be possible. For projects that are considering MUS in a geologic area with no prior managed recharge, the feasibility evaluation should analyze available data that can be used to evaluate the feasibility of recharge. It should also propose a course of investigation to assess recharge techniques. A phased course of investigation that spans literature review to field-scale testing is recommended. Field investigations and pilot testing for MUS in areas without prior MUS projects will typically have a greater scope and magnitude than in areas where MUS projects previously have been implemented.

Site selection may also be a major issue. Land ownership or nearby land uses may severely constrain the potential locations of recharge facilities. If, however, there is some degree of freedom in site selection, a location suitability assessment may be useful. An example of one kind of suitability assessment is given for ASR in support of the Florida Everglades restoration in Brown (2005). There, an index was used in which potential sites were ranked by weighting eight factors, including such disparate issues as ecological suitability, existing uses of the aquifer, groundwater quality, road density, access to power lines, and aquifer transmissivity.

Given the large number of site-specific variables that must be considered at the beginning of the process, decision trees may be a useful aid. An example of such a tool, in this case for selection of the most appropriate groundwater recharge technology (which overlaps Phases I and II), is presented in Figure 6-1.

As shown in the figure, the first critical question is what aquifer is being considered for use in the MUS system. If a confined aquifer is being considered, then direct recharge using wells is the only feasible alternative. Direct recharge may include either single-use recharge wells or the dual-purpose wells used in ASR systems. If the goal of a groundwater recharge project is to provide short-term storage and the water must be recovered quickly, ASR systems might be the only feasible alternative. If an existing distribution and well system may be utilized as part of an ASR system, then dual-purpose wells might be the best choice. If an unconfined aquifer is being considered, there are no constraints on the choice of recharge method.

For unconfined aquifers, one of the constraining variables on the choice of technologies is the depth to groundwater. As depth to groundwater increases, the cost of recharge wells increases. Depths ranging from 100 to 200 m have often been found to be a cutoff point at which recharge wells become more costly than surface recharge systems; however, site-specific factors such as land and drilling costs may make a different depth more appropriate (Bouwer, 2002). Therefore, the effect of depth should be evaluated for each situation, and land availability might make surface recharge basins more economical even with shallow groundwater depths. In situations where the depth to water is greater than 100 m, particular attention should be given to evaluating whether water recharged at the surface will flow down to the aquifer where it is planned to be withdrawn. If the evaluation indicates that a significant portion of the water recharged will not reach the aquifer in a reasonable amount of time, surface recharge may not be effective. This factor may be particularly important in arid areas where the depth to groundwater is commonly great.

The feasibility evaluation should also assess potential impacts on adjacent landowners, including consideration of potential changes in groundwater levels.

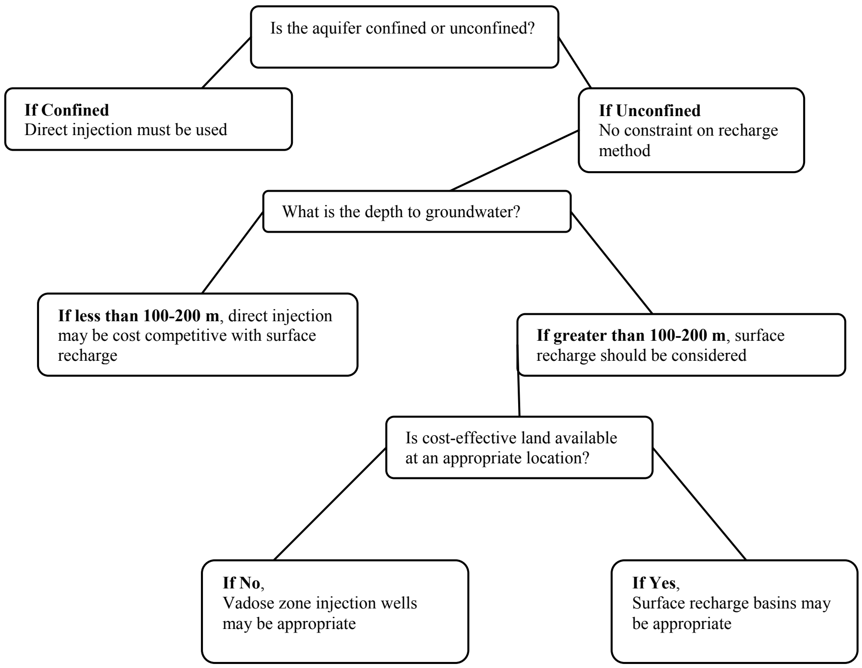

FIGURE 6-1 Sample decision tree for selection of groundwater recharge method. Another constraining question that must be considered is the availability of appropriate land. Land price and availability are key considerations. In addition, the location of the land and the cost of the distribution system to deliver water to the land are also important.

Such impacts could occur whether groundwater levels rise or fall, affecting existing water users or environmental resources (see Chapter 3). Projects should be designed to minimize the degree of impact to the maximum extent practical. For certain types of water sources such as stormwater, land might be required not only for groundwater recharge, but also as an aid to catch and store the water.

Governance is another important issue. Regional water authorities can be useful in implementing projects but can also create complicated interagency issues that can get in the way of solving water supply issues. In particular, the relationship between a proposed regional authority and participating local agencies needs to be carefully considered. For example, if a regional water authority is proposed, consideration should be given to the implications of a publicly elected local board delegating its responsibility for water supply development to an interagency regional water authority whose governing board may be appointed rather than elected. In such a case, local agencies may have concerns about a non-elected board setting policy.

The regional authority may also have different interests than the local agencies. The manner in which control and ownership of a project is assigned, rights to storage space and stored water are allotted, and related ownership and control issues are decided are important considerations. Thus, regional water authorities may have a useful role in implementing new projects, but the relationships among these authorities and local agencies should be assessed carefully.

As listed above, many other factors are involved in feasibility analysis, and this is simply an illustration of an approach that can be taken to guide one through what in the end is a very long and complex decision-making process.

Phase II: Components

The following text summarizes components typically addressed in an MUS pilot program. Note that these components vary slightly between surface spreading and well recharge systems. Monitoring issues are mentioned only briefly here; they are discussed in more depth later in the chapter.

- *Pilot program goals*—evaluate hydrogeology, including permeability, water quality, pressure or water table gradient, travel time, projected hydrogeological effects.
- *Soil borings*—evaluate the lithology, depth to groundwater, confining zones, aquifer materials, and aquifer properties; in addition to soil borings, core testing, split sampling, and side wall testing should be considered as some of the viable options to better determine aquifer characteristics. A lithologic log should be prepared and representative cuttings should be preserved when conducting soil borings or well drilling. Sieve analyses and mineral classification should be completed for the sediments recovered during drilling.
- *Tracer studies*—employ intrinsic or added tracers, tracer injection, monitoring locations.
- *Monitoring*—assess influent water quality, cycle testing for ASR, downgradient testing, native or background characterization (or upgradient).
- *Valve testing* (for ASR) —this should often be a preliminary review, with more detailed assessment after the range of flow rates is better defined.
- *Surface infiltration rates* (for surface spreading) —the infiltration rate as a function of time is a key parameter to determine feasibility.
- *Injection well recharge rates* (for ASR) —where the aquifer or associated aquitards are fractured or highly compressible, high injection pressures can cause hydrofracturing, and alternation of recharge and discharge can lead to irreversible subsidence.

- *Pretreatment evaluation*—determine the need for silt removal, pH adjustment, disinfection, et cetera.
- *Modeling evaluation*—predict impacts of supplemental recharge on gradients and travel time.
- *Risk Assessment*—see Monitoring section of this chapter for an example.

In the field testing program, it is important that the design of the investigation consider carefully the time and spatial scales used in the field testing compared to the full-scale MUS project. In most cases, the time and spatial scales used in the field investigations and pilot testing will be smaller than those of the full-scale project. Because of this, it is important to consider how field investigation results can be extrapolated for the full-scale MUS project. The manner in which the field data will be extrapolated should be an important consideration in the design of the field investigations and pilot testing. Key issues to consider in the design of the field testing include the following:

- How can the recharge water be conveyed to the field test site?
- How can the testing program be designed to collect enough data to characterize the spatial heterogeneity of the aquifer materials?
- What kinds of water quality transformations, positive or negative, during storage are likely? Adsorption, filtration, biodegradation, precipitation, dissolution, oxidation and reduction, and formation of disinfection by-products must all be considered. Changes in redox conditions as a result of recharging water are important to assess. When a new source of water is recharged, changes in redox conditions may occur that affect water quality. Particular attention should be given to redox potential changes and potential changes in concentrations of metals.
- What depths of the aquifer(s) should be monitored? Different aquifer depths may have different hydraulic properties and different water quality, thereby requiring monitoring at multiple depths.
- Is the well going to be subjected to corrosive environments? What type of well design should be used? What type of well screen and casing materials should be used? What size screen, how many screen intervals, and what formation intervals should be targeted?
- What are the rates of geochemical reactions that could occur between the recharge water, the aquifer materials, and the native groundwater?
- Over what length of time should the pilot testing take place to account for these geochemical reaction rates?
- In the case of brackish or saline aquifers, what are the salinity and the density of the groundwater, and what is the dispersivity of the aquifer at various distances from the well (see Brown, 2005)?
- How much field data have to be collected to build an accurate model of the full-scale MUS project?

- Overall, how can the testing program be designed so that it provides sufficient information to select the method of recharge, if more than one method of recharge is under consideration?

Overall, the availability of sufficient data for model input is important if the model is to be calibrated with an acceptable level of accuracy. Data should be collected early in the MUS model development and a sensitivity analysis performed to help determine any gaps that may be present in the data. In many cases, multiple, deep monitoring wells should be constructed to obtain the necessary data required for model calibration. However, the cost involved in doing so is quite high. Instead of executing several pilot studies in different areas, an alternative approach is to select one central area and perform detailed studies based on data from a well-equipped monitoring well. The drawback to this method is that the results may not be applicable across a large area if the system is heterogeneous. There is no easy answer to this issue.

It must be kept in mind that recharged water may mobilize some metals, inorganic compounds, and organic compounds bound in the aquifer material. The degree of mobilization will be impacted by a variety of factors, including pH, alkalinity, total dissolved solids, temperature, and the concentration of anions and cations already in solution. During operation of the pilot plant, core samples of representative subsurface strata taken in the vicinity of the boreholes could be obtained for use in soil or rock column tests to determine the likelihood of leaching or mobilization of chemicals. Native groundwater also could be collected during operation of the pilot plant for testing to determine any effects of mixing recharged water with groundwater.

A decision tree example for Phase II, in this case for treatment requirements prior to groundwater recharge to maintain hydraulic capacity, is shown in Figure 6-2. The type of treatment depends primarily on the choice of groundwater recharge method. For recharge basins, the removal of inorganic suspended solids is the major concern for maintaining infiltration rates. When vadose zone or recharge wells are used, infiltration rates may be reduced by suspended solids (especially critical for the former since there is no mechanism to backwash solids from the wells), biological fouling, or gas entrapment. Gas entrapment is not listed in the decision tree since the solution is often operational and is not a treatment issue.

Clogging is a dominant operational issue in MUS and is discussed in more detail both in Phase V (operation and maintenance) and in a separate section later in this chapter.

Phase III: Design

The following text summarizes components typically addressed in the design phase of an MUS system.

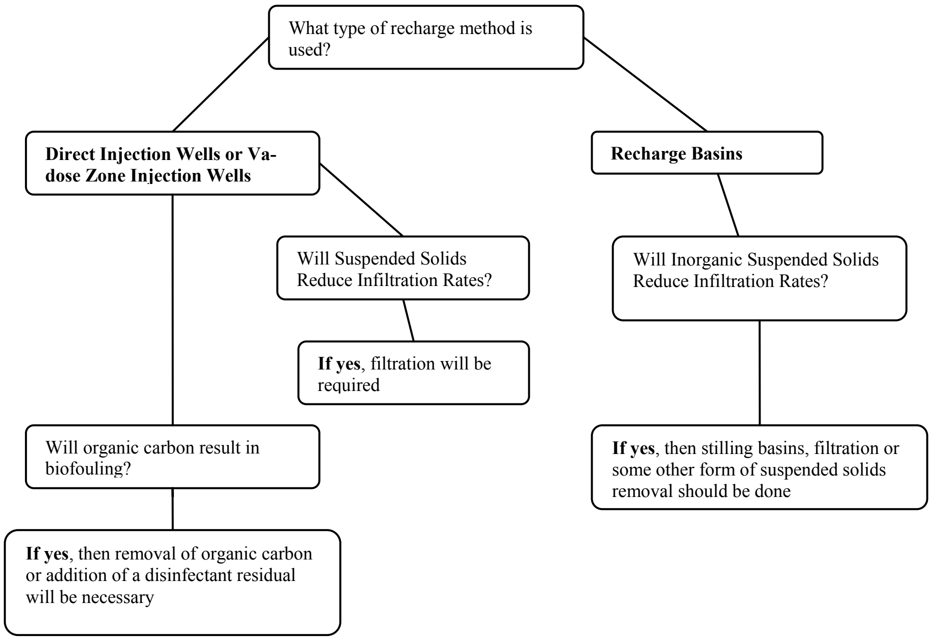

FIGURE 6-2 Decision tree for treatment requirements before recharge with respect to hydraulics.

Pre-design

Results from the field investigations and pilot testing should be evaluated thoroughly and utilized to prepare the project description. Assessment of the depth to groundwater is important to determine whether a vacuum situation could occur. The potential occurrence of a vacuum condition and air entrainment are important when considering the design of recharge wells.

The method of recharge and the amount of water to be recharged over defined time scales (e.g., per month, per year), should be defined in the project description. *Average and maximum recharge amounts* should be estimated.

The aquifer zones where the water will be stored and the duration of storage project description should also be defined.

Potential losses of water should be estimated.

The anticipated quality of the water after recovery should be specified.

The type of water treatment before recharge or after recovery should be determined if treatment is required. *Any unresolved issues* that need to be resolved prior to beginning design should be identified.

Additional field investigations should be conducted before commencing design if the unresolved issues are fundamental—for example, if there is significant uncertainty regarding how much water can be recharged or whether adverse water quality changes will occur after recharge. Potential interference among two or more recharge (or production) wells should be carefully investigated in design of an MUS system. This is an ideal area to use models, which depend on the quality of data on hydraulic conductivity, porosity, recharge or leakage, and so forth. Such models may provide fairly good guidance for well placement and discharge limits. For example, Finch and Livingston (1997) discuss the use of a groundwater model for managed underground storage using the La Luz well field in Alamogordo, New Mexico.

The pre-design phase typically concludes with preparation of a pre-design report that includes the project description and a description of the primary features of the project, including the facilities to be constructed and the estimated project cost.

Phased Design

Design should be implemented in phases to allow adding progressive levels of detail; as progressive levels of detail are added, the design should be reviewed rigorously and compared to the project description to verify that the design will allow implementation of the project. The design and operation of the MUS project should be integrated into the overall water management strategy for the area or region.

Rigorous review of the design at its initial stages is as important as, and in some respects more important than, review of the design at the final stage. Review in the early stages is critical to minimize subsequent redesign or inclusion of unnecessary or ineffective project elements.

If wells are to be constructed, careful consideration should be given to their design and construction. Installation of additional well sounding tubes and gravel feed tubes should be considered for gravel-packed wells. The drilling method should be matched to aquifer conditions. A proper, engineered drilling fluid program is important for drilling methods using drilling fluids. The manner in which well development is conducted after the well has been installed is important to its productivity. Well components such as casing, screen materials, submersible pumps, shafts, and bearings should be considered carefully with respect to corrosion issues. Epoxy-lined, stainless steel, and other materials should be evaluated as needed to minimize corrosion damage.

Engineering design of the system that needs to be constructed and specification of the operations and maintenance activities and their cost should be included. *An operations and maintenance program* should be developed. This should include a written operations and maintenance manual and should describe the water quality monitoring needed to assess the quality of the recovered water and any changes in water quality during recharge or storage that are important to understand. The operations and maintenance program should also account for the upkeep of recharge facilities and the cost of equipment replacement.

A key issue, if the project includes a water quality treatment system, is to include sufficient flexibility in the system so that it can be modified if conditions change and additional treatment is needed. If a system is designed to treat the water prior to recharge or after recovery, it needs to have flexibility to be modified if the influent water quality varies beyond the range anticipated during design or if new conditions occur. Flexibility should include the ability to add additional treatment units without having to remove existing units. Availability of space and adaptable piping and instrumentation are important factors to consider.

If the project is to be constructed in phases, it may be beneficial to construct the backbone water distribution, electrical, and instrumentation systems for the ultimate size of the project during one of the initial project phases. This can minimize the cost of adding future phases to the project.

Phase IV: Construction and Start-up

The following text summarizes components typically addressed in the construction and start-up phase of an MUS system.

- *Construction*—MUS facilities are sometimes constructed in environmentally sensitive areas; it is important that construction be implemented in accordance with relevant environmental laws and regulations.
- *Commissioning*—Generally speaking, this is the process of testing individual components that have been constructed to verify that they function properly; individual components are also tested to verify that they operate in conjunction with related components.
- *Startup*—After commissioning is completed, start-up is the process of operating the entire system to verify that performance criteria are met and the system operates reliably. If there is a permit with water quality limits or criteria, startup should include water quality monitoring to ensure that the permit limits are satisfied.

One key issue in the construction and start-up phase is to operate the system for sufficient time during start-up to ensure that it is reliably producing water according to permit conditions. If a construction project is near completion and over schedule, there may be pressure to shorten the commissioning and start-up phases to recover time in the schedule. In the event that the commissioning and start-up phases are not properly completed, there is an increased risk of producing water that does not meet permit conditions, which can result in loss of public confidence in the project.

Another important point is that the entire range of technical staff involved in planning and designing the project should remain involved with it. Since MUS projects often involve a range of technical disciplines, from water quality to engineering to geology, it is important to maintain the involvement of each discipline as the project moves forward. If changed conditions are encountered during construction, commissioning, or start-up, these changes should be communicated to each discipline.

Phase V: Operation and Maintenance

Some specific operational challenges identified in MUS systems are discussed below, with separate discussions for surface and well recharge systems.

Surface Spreading Operational Challenges

Surface spreading using recharge basins is the most common method for recharging untreated surface water or reclaimed water into MUS systems. Removal of clogging material that retards percolation is the main maintenance activity with recharge basins. Without removal of the clogging material, recharge basins can rapidly foul and become much less effective for groundwater recharge, as shown in Figure 6-3. The benthic clogging layer (BCL) can be a combination of inorganic and biological material. For systems recharging stormwater, the clogging material is typically composed of fine silts and clays that can form a layer plugging the surface of the recharge basin. Biological material, such as algae, bacteria, and organic detritus, can contribute to the clogging layer. A relatively thin layer (less than 2 cm) of fine (organic or inorganic) material is capable of significantly reducing percolation rates.

Control of weeds and vectors is also an important function. Weeds and surface debris interfere with maintenance of recharge basins and become a visual blight. Insect vectors and nuisances such as midges (chironomids) create problems in neighborhoods near surface spreading facilities. Control of both weeds and insects is an important part of operating spreading facilities. However, recharge agencies need to be cautious to avoid the use of persistent chemicals that might affect recharge water quality and therefore should focus more on mechanical methods and biological controls for weed and pest control.

Water agencies use several different methods to clean recharge basins. The simplest systems involve disking or ripping of clogging material to restore percolation capacity. This approach works best with shallow recharge basins that are routinely rotated between wetting and drying cycles and where silt loads are minimal. In these types of systems, biological clogging may predominate and the drying process is sometimes sufficient to restore percolation capacity by allowing the surface to crack and open up to the infiltration of water.

FIGURE 6-3 Clogging layer in recharge basin operated in Orange County, California, adjacent to cleaned portion of recharge basin where the clogging layer was removed. Photo courtesy of Adam Hutchinson, Orange County Water District.

Where silts and clays are a more significant factor in clogging, ripping and disking can have the adverse effect of driving fine sediments deeper where they may contribute to a long-term decline in percolation capacity that is more difficult to restore. For recharge basins where heavily silt laden stormwater is recharged, percolation capacity can be reduced very rapidly (Figure 6-4). Draining and scraping these basins is necessary to remove the silts and clays and restore percolation rates. After basins are drained or pumped dry, bulldozers and scrapers may be used to push the clogging material up the slope to the shoreline where it can be hauled away. Since the scraping process is imprecise, much of the removed material is sand. The sand can be recovered and returned to the basin bottom after washing and separation from the finer silts and clays. This sand washing and recovery process adds to space requirements and the maintenance and operations costs of spreading facilities.

With shallow basins, where depths are typically less than 10 feet, the draining and scraping process can be done relatively quickly and the basins can be returned to service without significant downtime. With deep basins, sometimes more than 60 feet in depth, the draining and scraping process is more lengthy

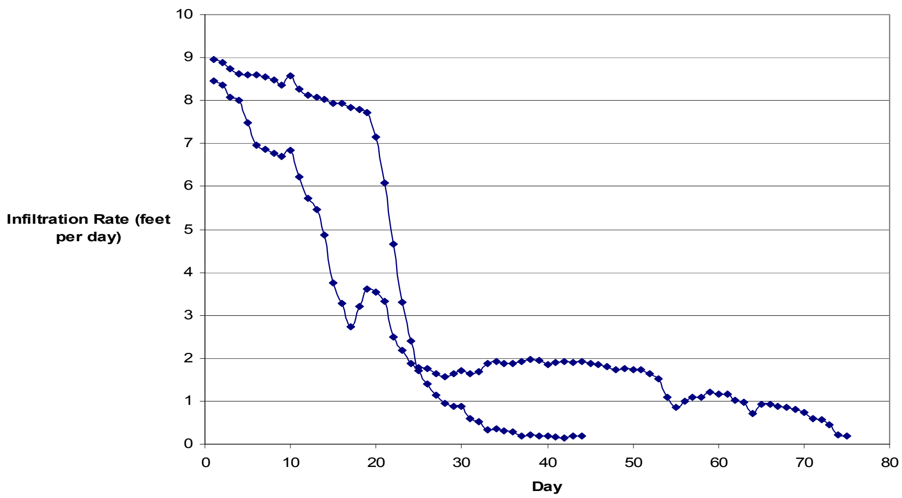

FIGURE 6-4 Recharge rate data from Kraemer Basin (Orange County, California). The decrease in recharge rate is due to formation of a clogging layer on the basin bottom, primarily as a result of fine-grained sediment transported with the recharge water. SOURCE: Reprinted, with permission, from Greg Woodside, Orange County Water District. Copyright 2007 by Orange County Water District.

and results in longer downtime, with the potential for greater water loss while the basin is out of service. Because percolation rates have slowed markedly by the time that cleaning is initiated, especially in the bottom area of a clogged basin, the basin may not drain naturally and pumping to transfer the water to other recharge basins may be required.

Wind agitation and varying water levels can provide a natural cleaning process for the sidewalls of some deeper recharge basins that are not amenable to more conventional cleaning processes. Abandoned gravel pits have been used in some areas for groundwater recharge, and very steep slopes preclude the use of surface scraping to remove clogging material. Drying of steep sidewalls can help restore percolation rates. Resting the sidewalls by lowering water levels works in the same way as the rotation and resting of shallow basins to restore recharge capacity. In deep basins, sidewall percolation may be much more significant than percolation through the basin bottom. While fine sediments tend to sink and clog the bottom most quickly, sidewall percolation can be sustained for much longer periods.

Because of the potential for greater sustainability of sidewall percolation, some recharge agencies have constructed recharge ponds that utilize a system of ridges with steep enough slopes to allow fine sediments to drop into the troughs while wind and wave action and water level changes naturally clean the slopes of the ridges. This approach helps to maintain percolation capacity for longer periods without draining and scraping to remove clogging material. The system also provides greater surface area for water to infiltrate.

Other innovative approaches are being tested by groundwater management agencies, including the use of submersible devices that disturb the clogging layer and pump out the fine sediments, leaving behind the coarser sediments that allow percolation. The basin cleaning vehicle (BCV) developed by Orange County Water District is shown in Figure 6-5. This type of system allows clogging material to be removed without interrupting the percolation process and avoids the water loss and expense associated with the downtime for draining and cleaning deep basins. Since heavy silt loads from storm flows can overwhelm the system, draining the basin and rehabilitating the basin bottom are still required on a periodic basis.

FIGURE 6-5 Basin cleaning vehicle (BCV) developed by Orange County Water District, California. This type of system allows clogging material to be removed while avoiding the water loss and expense associated with the downtime for draining and cleaning basins. Photo courtesy of Greg Woodside, Orange County Water District.

Since scraping the clogging material from the bottom of dried basins also results in the removal of clean sands, alternative approaches for removal of clogging material after drying are being evaluated. As clogged basins are dried, the layer of fine sediments on the surface cracks and curls up into chip-like structures. Certain types of beach cleaning equipment may offer the potential to remove only curled chips of clays and silts and leave the clean sands behind.

The greatest operating expense for recharge basins is the cost of removing the clogging material that retards percolation. For recharge of stormwaters, basin cleaning may be the only significant operating expense. Recharge basins are usually cleaned when percolation rates have declined to a point that groundwater cannot be effectively recharged. The overall capacity of the recharge facilities and the amount of water available for recharge will determine how quickly basin cleaning must be accomplished. In areas where seasonal storm flows can be anticipated, basin cleaning is generally done just prior to the storm season to facilitate capture of the maximum amount of stormwater.

Well Recharge Operational Challenges

Recharging with wells gives rise to specific challenges. One of the primary challenges is the rate at which wells can recharge and the decline in recharge rate through time at wells. Another challenge is controlling the rate of water flow into a well to prevent adverse flow conditions that will exacerbate the decline in the recharge rate. These issues are addressed in Pyne (2005), Segalen et al. (2005), and Brown (2005) and are summarized here:

The drilling method appears to be important to well productivity in many cases. For example, Segalen et al. (2005) found that production wells drilled using cable tool tended to outperform reverse circulation rotary drilled wells in the same formation. They also found that wells drilled using biodegradable mud gave rise to less clogging than when bentonite-based mud was used in the same formation and that residual mud near the borehole wall greatly limited recharge capacity. Finally, they concluded that wire-wrapped screens and natural gravel pack wells significantly enhanced well production relative to slotted casing and emplaced gravel pack in the same formation.

Clogging during recharge is as important for wells as it is for recharge basins and can be more difficult to overcome. In wells, clogging may be caused by physical factors such as suspended sediment or air entrainment, chemical factors that involve precipitation on the well screen or in the formation next to the well, or biofilms. Clogging is discussed in detail later in the chapter.

Cascading control is one of the more important components in recharge well design and operation. Cascading occurs when the water level in the recharge piping does not rise to ground surface during recharge. Allowing water to cascade down the well can lead to significant plugging problems due to air entrainment in the storage zone and induced geochemical or bacterial activity (Pyne, 2005). Cascading can also cause structural problems due to cavitation damage to pipes, valves, and fittings. To avoid these problems, water can be introduced into a well through the pump column, the annulus between the pump column and the casing, one or more injection tubes inside the casing, or some combination of these approaches. The following factors dictate selection among these alternatives (Pyne, 2005): casing diameter; static water level; type, size, and capacity of the pump; specific capacity and specific injectivity of the well, expected production rate; and range of recharge rates.

A careful management and balancing of recharge and recovery rates is important to the long term viability and integrity of the gravel pack. Compromising the gravel pack can result in unacceptable levels of fine sand or debris production, thereby increasing wear and tear of the components and limiting the life of the well.

Corrosion of the wells can be a major concern under certain conditions. These include low pH or high dissolved oxygen; hydrogen sulfide, chloride, or other salts; carbon dioxide; or temperature (EWRI/ASCE, 2001).

Finally, *recovery efficiency*—that is, the volume of water recovered as a percentage of volume recharged—is of critical interest, especially in areas where the aquifer salinity is fairly high or where the aquifer structure or geometry is complex. This topic is covered in detail in Chapter 3.

Phase VI: Project Review and Adaptive Management

Adaptive management is a key principle for the development, regulation, and operation of MUS systems. Assumptions made in the feasibility evaluation may have to be adjusted based on experiences during the pilot testing stage. New information acquired during operation and maintenance of MUS systems may result in continuing refinement and development of new approaches and technologies to minimize risk and increase the efficiency of recharge operations, improve water quality, or enhance the sustainability of the recharge and recovery of water stored underground. Such basic operational parameters as percolation rate (e.g., because of plugging of the aquifer), pretreatment (e.g., pH control to prevent aquifer pore space clogging from manganese release, and posttreatment (e.g., to prevent damage to water distribution systems or for aesthetics) may need adjustment.

However, adaptive management has to function at more than just the operational level. Even the regulation of MUS systems needs to evolve as more information is developed about the effects of introduced water on underground systems. Both regulators and MUS project managers need to reevaluate on a regular basis the effectiveness of existing procedures and regulations designed to protect water resources in light of the performance of MUS systems. For example, adaptive management may lead to changes in permitting requirements. This could mean reducing requirements because no impacts have been seen or a contaminant of concern has not been detected. It could also lead to new requirements if our understanding of an MUS system's dynamics has changed or because new contaminants have been identified based on trends and results of monitoring.

In some situations, establishment of an independent advisory panel can be useful to offer guidance and counsel regarding design, operation, maintenance, and monitoring strategies and parameters to ensure water quality integrity. It is best to have an independent third party administrate such a panel to ensure unbiased input and avoid conflict of interest implications. In addition to its role in optimizing operations, an independent panel can increase public acceptance of and confidence in the system (see "Public Perception and Involvement" later in this chapter).

As an example, under an agreement with California's Orange County Water District (OCWD), the National Water Research Institute appointed a panel of experts with various areas of expertise to assist OCWD in developing the Groundwater Replenishment (GWR) System program. The panel has met—and continues to meet—on a routine basis as the project has evolved. A report detailing the panel's findings and recommendations is prepared after each meeting and sent to OCWD and the California Department of Health Services. In this case, the panel was a requirement in the draft California Department of Health Services groundwater recharge regulations because of the high percentage of reclaimed water proposed to be recharged and it became a requirement of the GWR System permit issued by the California Regional Water Quality Control Board, Santa Ana Region.

A complementary exercise to the above would be a comprehensive status report approximately three to five years after the commencement of operations of an MUS project. Again, there should be considerable involvement of external assessors to contribute to the overall confidence of the community in the project.

Phase VII: Closure

Proper destruction of unused recharge wells or ASR wells is critical since abandoned wells in drinking water source aquifers can easily provide conduits for surface water or shallow groundwater contamination to reach deeper, normally more protected groundwater. Most jurisdictions have standards for proper well destruction to prevent vulnerability to migration of contamination from poorer-quality zones to higher-quality groundwater zones. In ASR systems where freshwater has been injected into saline or brackish water aquifers to create a freshwater zone, abandoned ASR wells also offer the risk of carrying higher-salinity water into freshwater zones if not properly sealed and destroyed.

Recharge basins or recharge ponds can readily be converted to other land uses if land values become too great to justify retaining large-scale spreading facilities. Ease of conversion will depend on the depth of the excavated ponds and the cost of fill to reestablish historical grades or elevations suitable for development. Subsurface recharge alternatives such as exfiltration galleries (analogous to leach lines for recharge water) or vadose zone wells may be employed to retain some recharge capacity on land that will be used for parking or landscaping.

The following sections explore in more detail four key operational issues associated with MUS. These progress from a very practical issue (clogging) to a scientific and regulatory issue (monitoring and indicators), a societal issue (public perception), and finally to financial considerations.

PREDICTION, REDUCTION, AND PREVENTION OF CLOGGING

Prediction

The broad spectrum of interrelated factors and the relatively frequent occurrence of clogging render it an important consideration during MUS planning, design, testing, and operation. Clogging may be predicted by bench-scale testing, multiscale field testing, indices calculations, and modeling. In a recharge basin for example, one of the most common clogging potential estimates involves use of infiltrometers, which are installed in the field to measure local infiltration rates. Results of laboratory column studies are particularly useful as well. Types of empirical clogging potential methods, such as the parallel filter index (PFI) column study, are summarized in Table 6-1. More sophisticated column studies can be designed to assess interrelationships among all clogging factors (Rinck-Pfeiffer et al., 2000), such as the dynamics of precipitation and microbial activity.

Limitations of the membrane filtration index (MFI), assimilable organic carbon content (AOC) and PFI methods have been described by Bouwer (2002):

> Experience has shown that MFI, AOC, and PFI are useful parameters for comparing relative clogging potentials of various waters, but that they cannot be used to predict clogging and declines in injection rates for actual recharge wells, which also depend on well construction and aquifer characteristics. Thus full-scale studies on recharge test wells are still necessary to determine feasibility and design and management criteria for operational recharge wells. Practical aspects such as a varying flow in the water-supply pipes to the recharge project and associated possibility of fluctuating suspended-solids contents in the water also play a major role in well clogging. The suspended-solids fluctuations can be caused by formation of biofilms in the pipelines during periods of low flow, and by erosion of the biofilms during high flow. Treatment of the water at the recharge site to remove suspended solids before well injection might then be necessary.

Biuk and Willemson (2002) also note lack of quantifiable reproducibility of results at the field scale; however, their preliminary study exhibits potential calibration of the MFI. By accounting for the mathematical relation between the MFI and aquifer media characteristics, a satisfactory correlation between expected versus observed clogging rates is observed for a limited data set.

TABLE 6-1 Selected Plugging Potential Empirical Methods

Clogging Predictor	Abbreviation	Method
Membrane filtration index	MFI	Describes suspended solids captured via microfilter in units of time per volume^2
Assimilable organic carbon content	AOC	Based on microbial growth in terms of carbon concentrations
Parallel filter index	PFI	Passing recharge water through columns filled with aquifer media, measured in flow rate per unit area
Bypass filter test	BFT	Passing recharge water through spun polyester cartridges while monitoring flow rates; allows for calculation of suspended solids

SOURCE: Olsthoorn (1982); Bouwer (2002); Pyne (2005).

Results of column studies, index calculations, and infiltrometer data can be scaled up to address field or operational conditions for recharge basins. However, the transfer of these data to full scale may require an intermediate step to validate the data in terms of heterogeneity and temporal variations. Bouwer (2002) suggests that test basins on the order of 30 m × 30 m should be employed to address this concern.

Numerical models can also be used to forecast clogging potential. A particularly robust method, Easy Leacher® 4.6 (Stuyfzand, 2002), predicts the accumulation rate and chemical composition of clogging sludge layers in recharge basins. While this application does not model the hydrologic clogging process, it does predict the rate of sludge accumulation and its composition. Moreover, the model allows for sensitivity analyses to assess optimal conditions to reduce sludge development. Because complexities of geochemical and microbiological processes are not fully understood and uncertainties exist when scaling up from laboratory to field scale, other numerical models exist to facilitate sustainable design and operation of MUS systems (e.g., CLOG; Perez-Paricio and Carera, 1998).

Reduction and Prevention

During MUS system design, clogging potential may be reduced by the addition of pre-treatment systems to remove suspended solids, nutrients, or microbes. In the case of a recharge basin, design of the basin floor, including sediment grain size and morphology (e.g., ridge and furrow; or flat surface), can also reduce compounding long-term effects of clogging.

Monitoring the effects of clogging is an important practice during MUS system operation. Of primary significance are observed reductions in infiltration or recharge rates. Other monitored parameters include water quality (e.g., dissolved oxygen, pH, TDS) and rates of change in hydraulic head within recharge zone monitor wells. The bypass filter test (BFT) and PFI methods can provide early warning regarding the onset of clogging. In addition, video analysis of the borehole and well screen (i.e., biofilm or precipitate buildup) is also a useful method.

MUS system maintenance can be categorized as either physical-mechanical or chemical. In recharge wells, physical maintenance practices often include backflushing (e.g., backwashing or redevelopment) or some other form of physical agitation to loosen and remove plugging materials such as (1) compressed air jetting, (2) controlled sonic blasting, and (3) pressurized CO_2 injection (liquid and/or gas). Other physical methods could include brushing or swabbing the screen. Backflushing practices can be optimized by establishing the relation between injection rates, total suspended solids, and backwashing frequency. It may be necessary to monitor the quality of the water produced during maintenance and construct storage systems to contain it. Physical maintenance in a recharge basin generally includes breakup and/or removal of the low-permeability "cake" layer. Breakup practices include disc harrows or rotary tillers, or a "dry-and-crack" technique. Several methods exist with respect to cake removal, such as scraping; however, care must be taken to minimize compaction of the basin floor.

The goal of chemical additives is generally to dissolve clogging constituents. Heat may be used to augment the process. With regard to biomass buildup, chlorine or related chemicals can be added to recharge wells; however, this is weighed against the potential for formation of disinfection by-products such as trihalomethanes. Adjustment of pH during recharge and borehole acidification is among the techniques used to remove precipitates. In addition, these practices may be complemented by physical well agitation to optimize removal of clogging material. Flocculation and removal of swelling clays is also accomplished by chemical additives. It is noteworthy that changes to the borehole environment with respect to acid-base or redox reactions may adversely affect the quality of the stored or recovered water. As such, awareness of the potential hydrogeochemical reactions and subsequent monitoring of constituents of concern is an important consideration.

For more detail on MUS well clogging issues, texts such as Mansuy (1998), Bloetscher et al. (2005), and Pyne (2005) can be consulted.

MONITORING ISSUES

Monitoring is an integral part of MUS site selection, design, and operation. Successful MUS involves careful and thorough project-specific assessment that includes chemical and microbiological monitoring to document system performance and evaluate the reliability of the process. There are several roles for monitoring, as outlined below:

- Establish the feasibility of the site by characterizing the hydrogeology, and identify pertinent water quality issues. Knowledge from this phase of monitoring is used to develop a site conceptual model.
- Obtain parameters for design and operation, such as recharge and extraction well placement, hydraulic capacity and recovery, and appropriate travel time or residence time.
- Determine the need for pre- or posttreatment of the water, such as removal of particles and biodegradable organic matter from the source water or removal of excessive dissolved constituents in the extracted water.
- Comply with regulatory requirements.
- Document the performance to build trust with consumers and improve public perception. Monitoring provides an opportunity to become proactive for emerging contaminants and issues.
- Adjust system operation in the future in response to what has been learned from ongoing monitoring (i.e., part of adaptive management).

A number of the roles for monitoring identified above are addressed elsewhere in this report. For example, the need for information on the hydrogeologic and water quality parameters is addressed in Chapters 3 and 4. The nature of the present regulatory framework is mentioned in Chapter 5. Earlier sections in this chapter cover some of the important factors in design and operation and describe the benefits of adaptive management and risk management. The sections below elaborate on three remaining issues pertinent to monitoring. The first is *where* to monitor? The second is *what* should be monitored? The final topic is the *frequency* of monitoring.

Where to Monitor?

The subsurface has the capacity to remove or attenuate many chemicals and pathogens, thus improving the quality of the source water. A monitoring program is needed to document the water quality behavior and establish the reliability of the MUS system. This will involve installation of monitoring points to track the behavior of the water and the constituents in the water as the source water is introduced, stored, and eventually extracted. A number of reports on the topic of monitoring well installation and networks for characterizing subsurface processes exist for the interested reader (NRC, 1994, 1999, 2000, 2003).

For ASR, it is recommended that several wells in addition to the ASR well be monitored to establish the physical extent of the introduced water and help pinpoint the recovery efficiency. The exact number of monitoring wells will vary with their purpose and the degree of existing knowledge of the site, but in all cases dialogue with regulators beginning with the inception of a project is recommended. Since federal and state agencies are charged with protecting an entire aquifer for all users, when an MUS project is proposed, they will wish obtain information on the impacts over a broad area of an aquifer, not just the "bubble" around the ASR well. Since it is unlikely that the physical or geochemical behavior of the stored water can be accurately assessed or modeled using information from a single monitoring point, the requirements of science and regulatory authorities are not necessarily in conflict.

For systems that involve recharge at one point and extraction at a downgradient point, it is also recommended that several monitoring wells be installed downgradient from the recharge site prior to initiation of recharge to document changes in the quality of the water as a result of mixing with native groundwater. Placement of such monitoring wells is also advantageous for confirming estimates for travel time from recharge wells to existing and proposed extraction wells. The depth interval(s) in monitoring wells for MUS systems should relate to the depths of injection zones or other important hydrostratigraphic units.

In areas where the aquifer properties, groundwater velocity, and groundwater quality are poorly understood, multiple monitoring wells are commonly needed. The number of monitoring wells required also generally increases with the areal size of the project and the likely risk to other users of the aquifer. In karst aquifers, it may be necessary to utilize geophysical methods to characterize hydrogeologic conditions. Unless geophysical methods or other detailed studies are conducted, there may be considerable uncertainty regarding the direction and rate of movement of stored water in karst systems.

The point of compliance for many regulatory programs is the location at which the source water is first introduced into the subsurface. This means that the source water must meet all regulatory requirements prior to introduction and storage. Such antidegradation approaches can in some cases be overly restrictive. They do not provide credit for improvements in water quality that can occur while the source water passes through the subsurface or resides in the storage zone. As noted in Chapter 5, there is precedent for balancing the benefits of the MUS project against a strict antidegradation policy. The content of Box 5-3 provides some language from the State of California that offers an opportunity to weigh the benefits of subsurface storage and water quality improvements against an antidegradation policy. Serious consideration needs to be given to allowing flexibility in the regulatory framework so that the point of compliance can be at the extraction well or downgradient monitoring well if the extent of water quality improvements merits such a designation and human health is not compromised.

What to Monitor?

Chapter 3 discusses several basic ideas for what needs to be monitored in the context of recovery efficiency. The degree of mixing between the introduced source water and the native groundwater can be assessed using a tracer test. The hydraulic properties of the subsurface are gleaned from step-drawdown pump tests. Cycle test monitoring is used to determine the recovery efficiency and look for potential water quality changes that occur after mixing.

For many types of recharge operations, water quality monitoring requirements are relatively limited. Most systems recharging drinking water through ASR wells or river water through channel beds or recharge basins into underground storage are essentially unregulated. Water quality monitoring requirements are therefore very limited. In some areas of the country, water deliberately introduced into the subsurface must meet water quality objectives that protect beneficial uses of the groundwater. The most restrictive use is typically for drinking water. Monitoring to ensure compliance with drinking water standards in the extracted water is therefore often the most basic requirement, regardless of the source of recharge water. So a frequent answer to the question about what to monitor is the list of contaminants that have drinking water standards. As noted in Chapter 4, a change in redox conditions in an MUS system can impact water quality. It is important to consider monitoring redox indicators and inorganics such as manganese, arsenic, and other trace metals that might be mobilized during MUS operations.

For waters of more impaired origin, such as reclaimed water, urban runoff, or agricultural runoff, there may be additional contaminants of concern in the recharge water. Examples include pharmaceuticals and personal care products (PPCPs), hormones, and other trace organic chemicals. These are usually called emerging contaminants (principally trace organic compounds of anthropogenic origin and may be better labeled trace organics), and most are presently unregulated. Chemicals that interfere with endocrine systems of humans and wildlife are termed endocrine disrupting compounds (EDCs). Chemicals that elicit a pharmaceutical response in humans are termed pharmaceutically active compounds (PhACs). EDCs and PhACs are not mutually exclusive classifications, because some, but not all, EDCs are also PhACs. Thousands of compounds have been reported to show endocrine disrupting properties, primarily in relation to estrogen effects (Global Water Research Coalition, 2003a), and more than 60 PhACs have been identified that impact the endocrine system of animals or humans in nanogram per liter or lower concentrations in the ecosystem. PPCPs comprise a very broad, diverse collection of thousands of chemicals, including prescription and over-the-counter drugs, fragrances, cosmetics, sunscreen agents, diagnostic agents, and many other compounds.

PPCPs and EDCs are found in many watercourses, usually at extremely low concentrations. In one 1999-2000 survey of the occurrence of organic contaminants in the United States, for example, samples collected from 139 streams in 30 states tested for 95 pharmaceuticals, personal care products, and known or potential endocrine disruptors found that 80 percent of the streams sampled contained at least one of the chemicals (Kolpin *et al*., 2002). Although measured concentrations were generally low and rarely exceeded drinking water guidelines, drinking water advisories, or aquatic life criteria, many of the compounds have not been subjected to toxicological testing to establish drinking water limits.

Unfortunately the information on occurrence of unregulated chemicals is much greater than the understanding of their significance. Several lists of emerging contaminants have evolved on the basis of new analytical techniques that have made it possible to examine extraordinarily low levels of compounds used in everyday life. We are essentially at the point of testing because we can, without knowing whether there is any significance to the findings. There are compounds with ecologic significance that may also have human health significance, such as some of the EDCs. There are many other compounds such as commonly prescribed or over-the-counter pharmaceuticals whose detection at parts-per-trillion levels may have no significance to either wildlife or humans. Nevertheless testing for a growing array of these compounds continues. Their detection by one researcher often prompts testing by others. As lower and lower detection levels are pursued, this problem of detecting compounds such as PPCPs and EDCs without understanding the significance of the findings becomes more serious. One of the most difficult questions facing both regulators and groundwater management agencies is the appropriate testing requirements for such compounds including detection levels. Additional information is needed on the levels of PPCPs and EDCs that are of potential health concern. Once those levels of can be determined, appropriate detection levels and test methods can be developed and potentially included for MUS systems using waters from impaired sources.

In the interim, some states such as California have established guidance levels for some of the compounds for which maximum contaminant levels (MCLs) have not been established. *N*-Nitrosodimethylamine (NDMA) is an example that is described in Box 6-1. As of 2006, the California Department of Health Services had established notification levels and response levels for roughly 40 compounds, and the list may continue to expand as additional chemicals are evaluated for guidance. The concept behind the testing is to develop information on the occurrence of representative compounds that may be used to guide future regulations regarding treatment requirements for indirect potable reuse. The notification levels and response levels are based on potential health effects, but without the comprehensive review necessary for establishment of MCLs. Some of these compounds will eventually be regulated with MCLs, but many will have only guidance levels for many years. Where waters used for MUS are derived from more impaired sources subject to a wider range of contaminants than more protected surface waters, monitoring for compounds with such guidance levels may be appropriate since these compounds are targets of concern in drinking water supplies.

BOX 6-1
NDMA Testing in Waters for MUS

N-Nitrosodimethylamine (NDMA), a highly toxic compound, has been found in various types of water, including reclaimed water recharged into aquifers used for potable reuse. NDMA is one of the few unregulated trace organic chemicals with toxicological data indicating risks at part-per-trillion levels. Under some conditions, NDMA has been found to be persistent in groundwater and therefore a potential threat for MUS systems. NDMA is a contaminant in some industrial wastewaters and is a disinfection by-product, particularly with chloramine disinfection.

Because of these factors, NDMA is a particular concern for recharge of reclaimed water and should be included in monitoring programs to verify suitable water quality for MUS systems intended for drinking water supply. Although the federal government has yet to develop regulatory limits for NDMA in water, the State of California has developed a notification level and a public health goal for the compound. In the context of other exposures to NDMA, through foods, beverages, and rubber and plastic products, NDMA exposures at part-per-trillion levels in drinking water may not be significant, but in the context of regulatory limits for other compounds in drinking water, testing and control of NDMA in waters for MUS appears appropriate.

There is an inherent conflict—both in time and money—between the desire for complete and comprehensive information and the need to keep costs reasonable and commensurate with risks. This raises the question, If one cannot monitor everything everywhere, continuously, and forever, how can one feel confident that the risks are manageable at a reasonable cost? One approach to answer this question is to develop a biomonitoring system to signal the presence of toxicity. A second approach is to employ surrogates and indicators for the many compounds and microorganisms of interest. Inorganic chemical analyses are not unduly expensive, and the demand for inorganic chemical indicators has therefore not been overwhelming. There are cases where conductivity or chloride is used as an indicator of salinity. Most of the need, however, has been to understand the risks from organic compounds and pathogens. A third, and often complementary, approach is to implement a risk management system. The following sections focus on online biomonitoring and surrogates and indicators for trace organics and microorganisms, and minimizing risk within MUS systems from source to supply.

Online Biomonitoring

Toxicological data have yet to be developed for many of the EDCs and pharmaceuticals found in water. A National Research Council (NRC, 1998) report on potable reuse, recommended development of fish biomonitoring methods to address unidentified chemicals and lack of toxicological data for identified chemicals. At least one study has been conducted to evaluate online biomonitoring methods (Schlenk et al., 2007) using Japanese *medaka* exposed to river water after recharge under ambient conditions in flow-through systems, which are subsequently examined for tumors and other anomalies. This online biomonitoring procedure, while promising, is not yet developed to the point where it can be implemented to evaluate the safety of potable water.

Surrogates and Indicators for Trace Organics

Traditional measures of organic matter, such as biochemical oxygen demand (BOD), chemical oxygen demand (COD), and total organic carbon (TOC), have been used as measures of treatment efficiency and indicate the presence of wastewater in a water supply. This in turn can signal the likelihood that specific trace compounds of health concern are present. TOC is composed mainly of natural organic matter, organic chemicals of anthropogenic origin, and soluble microbial products generated during biological wastewater treatment from the decomposition of organic matter (Drewes and Fox, 2000). The contributions can vary depending on location and season. Different approaches have been proposed to distinguish between naturally occurring and wastewater-derived organic constituents using differences in functional groups, structural properties, molecular size distribution, aromaticity, reactivity, or acid-base solubility (Drewes et al., 1999; Leenheer et al., 2001; Leenheer, 2003; Müller and Frimmel, 2002; Her et al., 2003). These approaches are promising and provide more insight into the origin of organic matter, but they often are semiquantitative and require a high degree of expertise for proper assessment. Further, TOC (and other bulk parameter) measurements are not a useful predictive tool for tracking the behavior of very low levels of some health-significant chemicals, and identification of one or more constituents in water that can be used as surrogates for unregulated chemicals is needed.

Analytical techniques such as ELISA (enzyme-linked immunosorbent assay), gas chromatography-mass spectrometry (GC-MS), and liquid chromatography-mass spectrometry (LC-MS) are available to identify and quantify individual trace organic chemicals, and quantitative structural activity relationship (QSAR) and quantitative structure activity-property relationship (QSPR) models have been used to predict the behavior of EDCs, PPCPs, and other chemicals. Such advanced analytical techniques require highly trained chemists to operate equipment that is expensive to purchase and use and, thus, are used mainly by university researchers and a limited number of agency and commercial laboratories. The growing number of trace organic constituents, particularly EDCs and PPCPs, make it impractical to routinely test water for the entire suite of known or suspected constituents of concern.

The lack of adequate indicators of surrogates is particularly important to MUS where the extracted groundwater is to be used as a potable supply. Some chemicals that exhibit unique characteristics or are poorly removed during engineered treatment or soil aquifer treatment (SAT), such as carbamazepine, may not be readily identified or quantified through the use of indicators or surrogates and specific testing for these compounds may be needed. Because of to their chemistry or behavior, other constituents are amenable to evaluation via indicators or surrogates. Clearly, there is no single constituent, indicator, or surrogate that is representative of the vast array of trace organic constituents present in recharge waters, and monitoring will have to include a suite of parameters. The selection of which specific chemicals, indicators, or surrogates to monitor is dependent on several factors, including the following:

- Type of recharge water (e.g., stormwater, river water, reclaimed water)
- Treatment prior to recharge, if any
- Type of recharge (i.e., direct recharge or surface spreading)
- Regulatory requirements (e.g., drinking water standards, antidegradation requirements)
- Specific trace organics known or suspected to be present in the recharge water
- Known toxic chemicals not amenable to detection or quantification by indicators or surrogates
- Time lapse between sample collection and completion of analyses
- Validity of analytical techniques used and confidence that the suite of parameters measured is indicative of water quality

In recognition of the need to identify appropriate indicators or surrogates for organic constituents, several research efforts have been undertaken in recent years. One promising approach has been advocated by Drewes and Dickinson (2007) and others to target the presence and concentration of many individual or types of organic compounds having known or suspected health significance using indicators and surrogates. In this case, an indicator is defined as an individual chemical occurring at quantifiable levels that represents certain physicochemical and biological characteristics of a family of trace constituents and provides a conservative assessment of removal (e.g., ibuprofen, NDMA), while a surrogate is defined as a quantifiable change of a bulk parameter that can serve as a measurement of the performance of individual unit processes or operations regarding their removal of trace compounds (e.g., change in TOC or conductivity through a treatment process).

The proposed methodology entails identifying several "treatment bins" (biodegradation, chemical oxidation, physical separation, etc.) into which chemicals are listed as to their removal. Removal of surrogates such as biodegradable organic carbon (BDOC) can then be compared to the removal of the various constituents of concern, and where the removal of surrogates corresponds to the removal of constituents or classes of constituents, the surrogates can be used for monitoring purposes.

Another approach is to establish a priority list of chemicals for monitoring. In recognition that monitoring the entire spectrum of potential EDCs in water and wastewater would be cost-prohibitive, the Global Water Research Coalition

(2003b) developed a targeted list of EDCs that would provide a basis for credible analytical determination of EDCs in water. It is understood that the priority list of EDCs is dynamic and additions or deletions to the list may be made as additional information becomes available.

The advantage of the above approaches is that easily measured bulk parameters can be used to simplify the trace organic analytical monitoring effort and provide a conservative assessment of removal. The disadvantages are that the indicator occurrence pattern may change, indicator selection requires regular review, and operational conditions determining removal can change over time. While the methodology used by Drewes and Dickinson and the Global Water Research Coalition (and similar methodologies being developed by others) may eventually prove to be appropriate for MUS systems using either wells or surface spreading, further evaluation and refinement is needed to validate the concept in practice. Using targeted indicators and surrogates to evaluate water quality and safety in lieu of intense monitoring for the plethora of unregulated organic constituents potentially present in water is a reasonable and realistic goal that is achievable with our current state of knowledge.

Microbial Indicators

The quality and safety of drinking water and groundwater have always been measured via fecal indicator organisms and in some cases the presence of viruses and other surface water associated pathogens such as *Cryptosporidium* and *Giardia* Environmental Protection Agency (EPA) proposed groundwater rule). As already mentioned, the nature (perceived as protected) of groundwater and its use (as a potable supply) dictate the absence of “indicator” microorganisms. Indicator organisms most commonly used include total coliform bacteria and *Escherichia coli*. These are a part of the regulatory targets for drinking water; however they are now known to have disadvantages and cannot be used as broadly as once intended. Fecal indicator bacteria are generally harmless themselves, but are found in high numbers in the gut of humans and other warm-blooded animals, including birds. These are excreted daily in the feces of people and mammals. It should be noted that total coliforms are found in soils and are generally used as a disinfection process control target and have yet to be associated directly with pathogens or human health risks.

- Fecal indicator bacteria including *E. coli*, enterococci, and virus indicators such as coliphage are also found in many environments, such as sewage (even treated sewage), septic tank effluent (liquid from a septic tank), septage (solids from a septic tank), manure and animal waste lagoons, and bird and other animal droppings. Heavy rainfall can wash the fecal wastes and associated indicator bacteria into nearby water bodies.

- Fecal indicators can be found in most waters and the indicator levels generally reflect the amount of fecal pollution. However, even in pristine waters there is a background level of fecal indicators. They are presumed to be absent from groundwaters.
- These fecal indicators are used to indicate the potential presence of pathogens, microorganisms that come from the gut and cause diseases such as diarrhea.
- Disadvantages and limitations in the use of these "indicators" includes the fact that that sources of fecal contamination cannot be determined with routine methods; regrowth of the fecal bacterial indicators occurs, and there is a poor relationship of the indicators to the presence of viruses, parasites, *Legionella*, and cyanobacteria.

The quality and safety of drinking water and groundwater have always been measured via these fecal indicator organisms, with MCLs set only for total coliforms and *E. coli.* Viruses and other surface water-associated pathogens such as *Cryptosporidium* and *Giardia* have been addressed through treatment technology rules based on removal and inactivation associated with filtration and disinfection (EPA long-term enhanced surface water treatment rule and ground water rule).

All surface waters will have some level of algae, bacteria, and parasites in them, and with increasing sewage inputs there will also be enteric viruses and other microbes of fecal origin. Thus, unless this water is pretreated to drinking water standards or infiltration systems are used to effectively remove some percentage of the microorganisms, the source or stored water will contain these microbes. The native groundwater could also contain some bacteria (*Legionella*) and protozoa (*Naegleria*) that pose a risk to human health (outbreaks and associated deaths have occurred for both of these microorganisms due to the use of groundwater; see Appendix A). The targeted microbial contamination level associated with acceptable risks would depend on the use of the recovered water. For potable purposes a maximum contaminant level goal (MCLG) of zero is the target for those pathogenic microorganisms. Finally, regrowth of bacteria and the free-living protozoa can occur depending on the conditions, but more importantly, attenuation (usually due to inactivation of bacteria, parasites, and viruses) occurs. For enteric viruses and protozoa, long-term survival is of concern and interest. As a part of the attenuation via filtration or dilution (diffusion) the concentrations of the microorganisms that may migrate and be transported into other aquifers has also been an area of research.

Monitoring for the wide range of microorganisms in source, stored, and recovered water has not been widely implemented. Thus, there is often a presumption of microbial water quality based on the monitoring of selected "indicator" species. As mentioned, the primary research has focused on drinking water MUS systems, thus, those microbes associated with fecal pollution and standards and rules for potable water have been the target of most of the controversy and studies. Bacterial pathogens are rarely monitored, a select group of viruses may be monitored on occasion, and protozoa are monitored in surface waters but not groundwaters. None of these groups of microbes are monitored in reclaimed waters on a routine basis (the exception being in the State of Florida, which requires monitoring of *Cryptosporidium* and *Giardia* in reclaimed wastewaters albeit at a low frequency).

A full description of the indicator bacteria (coliforms and alternative indicators such as enterococci and coliphages) other pathogenic bacteria (such as *Legionella*, *Arcobacter,* and Cyanobacteria), parasites (*Cryptosporidium* and *Giardia*; free-living amoebae), and enteric viruses (norovirus) is found in Appendix A.

Several key scientific data gaps have been identified that hinder the monitoring of microbial water quality in the assessment of MUS as well as other groundwater projects:

1. The type of microorganism to be used in studies. Most studies have used laboratory strains, and the survival rate is questionable as it relates to either less or more resilient groups of pathogens or naturally occurring fecal indicators on which monitoring programs may be focused. In addition, better surrogates of pathogens may be needed. Some have found, for example, that PRD-1 survival may be a good model for that of hepatitis A in groundwater (Blanc and Nasser, 1996), and PRD-1 has been used as an indicator of virus transport and as a resilient tracer in field studies (Harden et al., 2003; Paul et al., 1995; Ryan et al., 1999).
2. The influence of the native microflora in surface and groundwaters on the inactivation rates of fecal indicators, along with redox conditions and nutrients;
3. The impacts on fecal microbial survival of infiltration into aquifer environments conducive for storage and inactivation associated with pore waters; and
4. *In-situ* studies in general, because most work has been done in the laboratory.

A Risk Assessment Approach

Each MUS system has associated risks of physical, chemical and biological hazards. In a stormwater recharge basin, for example, hazards include spills, floods, or land-use changes that impact stormwater quality. The degree of risk is related to the likelihood and consequence of the hazard, or combination of hazards. Identifying hazards and assessing risk are important toward development of a sustainable water resource for the end-user or the environment. An example of this assessment process is the Hazard Analysis and Critical Control Point (HACCP) plan implemented for a stormwater to drinking water project in

Salisbury, Australia (Swierc and others, 2005).[1] The project employs ASTR as the final treatment step within a larger system that includes stormwater catchment/management, surface storage and cleansing wetlands to produce potable water (Figure 6-6).

Rather than monitoring at the end-point of the system, hazards and controls are evaluated along the entire system flowpath. The *hazard analysis* component of HACCP is based on a verified understanding of the processes, and identification of potential hazards and preventative measures. With this knowledge, each step along the process is assessed as a potential *critical control point (CCP)*, which is defined as a point, step or procedure "… at which control can be applied and is essential to prevent or eliminate a hazard or reduce it to an acceptable level." For example, a disinfection system or a water quality monitoring station is a CCP. Each CCP has associated hazards that may be unique to that step, and each is examined to identify the following requirements: 1) monitoring: allows for tracking of the operation and trend analysis to flag potential loss of control, 2) corrective action or response: in the event of loss of control, and 3) data/record management: evidence of adherence to procedures and events reflecting loss of control. These requirements are assessed in the context of critical limits (e.g., EPA MCLs) established for each CCP. Once these limits are established, monitoring requirements and corrective action procedures are developed that are specific to each CCP.

With regard to the ASTR CCP in this particular system, aquifer characterization as well as physical, chemical, radiological and microbiological processes are identified and considered in relation to variability in quality of input waters from the wetland treatment (reedbed cleansing) step. Contaminant attenuation can be modeled along the ASTR flow path, recognizing that the aquifer has finite sorption capacity. The modeling facilitates system understanding, including travel times along the flow path, which helps in the effective design of a monitoring plan. Monitoring wells are placed and sampled to allow performance tracking of the system; considerations for water-quality sample parameters and frequency are discussed in other sections within this chapter.

How Frequently to Monitor?

In most instances, the frequency of monitoring will be dictated by the regulatory jurisdiction that is overseeing the project. An efficient monitoring program is one that involves a frequent sampling schedule at the start of operation to develop a historical record of the hydraulic characteristics and water quality trends. As clear trends in performance emerge along with consistent sampling results, the monitoring frequency can be decreased with confidence. A frequent sampling schedule in the early stages of operation will help to build trust of consumers and improve public perception of the project. An early, frequent

[1] This section modified from Swierc and others (2005).

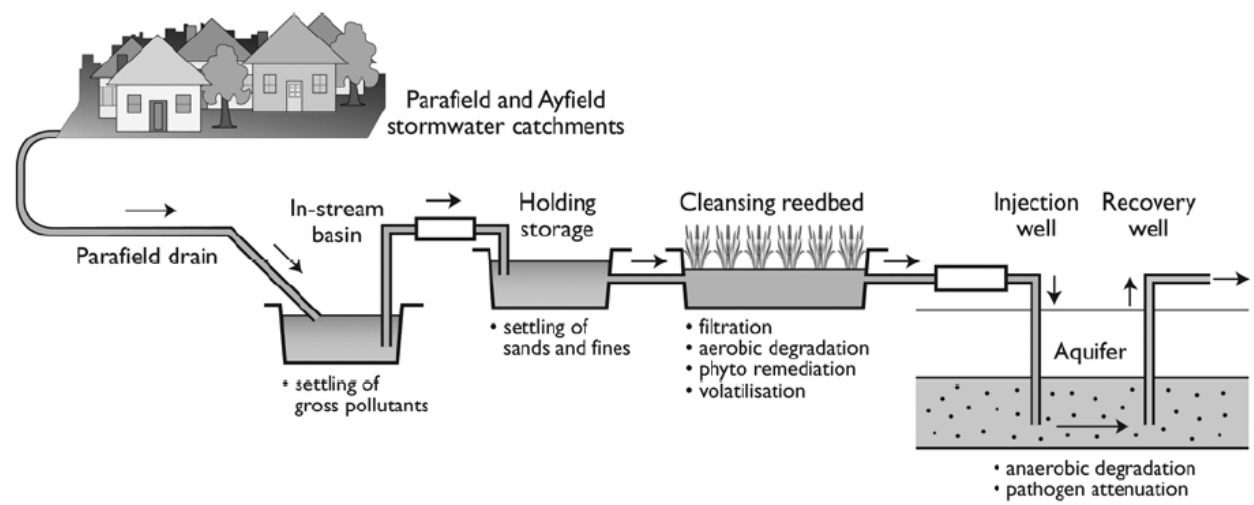

FIGURE 6-6 ASTR project site and scope of the HACCP plan. Available online *http://www.clw.csiro.au/publications/technical2005/tr20-05.pdf*. Accessed December 19, 2007. Reprinted with permission from CSIRO. Copyright by CSIRO.

monitoring program can document that the system is performing as intended. To the extent possible, predetermined decision rules should be established prior to commencing monitoring. The predetermined rules should reflect the goals of the monitoring program and provide criteria for the constituents to monitor, initial monitoring frequencies, when monitoring frequencies should be increased or decreased, and how results from the monitoring program should be acted upon.

Proactive monitoring combined with the ability to adapt the system is important to maintain a successful MUS project. Water quality is an evolving science and unknowns exist. For example, the manager of an MUS system needs to pay close attention to the range of trace organic contaminants that are candidates for monitoring. An ongoing water quality plan that takes measures to increase knowledge risks (e.g., occurrence of emerging contaminants) and to improve quality over time helps to gain public trust for an MUS project.

PUBLIC PERCEPTION AND INVOLVEMENT

As Seerley (2003) has pointed out, public involvement in MUS projects is not automatic—projects tend to be technically complex and regulatory processes not entirely clear. Nevertheless, although underground storage of water has not been a high-profile issue in many communities in the United States, public education and involvement constitute an important step in any type of water management undertaking. The general public has legitimate interests in, and concerns about, the quality and reliability of its water supplies. Any plans to initiate or enlarge a water storage project should consult with and educate the public in the process.

As public trust in public agencies and private corporations has declined over the past half-century in the United States (Pew Research Center, 2001), there is a possibility that the public might react negatively to projects and decisions on which there has been insufficient information and consultation. When some event or development regarding a water storage or treatment project does gain public attention, a variety of organized interests can be expected to engage in a struggle for media attention and public support (Seerley, 2003), widening the scope of involvement in decision making. Failure to engage the public at the outset of a project planning process may reinforce public mistrust when plans are publicized later. Under those circumstances, project planning and the decision-making processes can progress into competition for public opinion among organized interests. Loss of public trust in water providers and/or regulatory agencies can influence public opinion on MUS and other projects for a long time (Seerley, 2003). Trust takes time to build, can be lost overnight, and is especially difficult to restore.

When public outreach, education, and involvement have been taken seriously and pursued conscientiously, there are notable success stories of public engagement with water management activities. From decades of experience and research, some general principles have emerged for public participation in water management decision making. The Water Environment Research Foundation (WERF) published a useful review of the state of knowledge on this topic (Hartley, 2003). Although the WERF review was undertaken with particular reference to storage, recovery, and reuse of treated wastewater, its conclusions and recommendations are applicable to a much broader range of projects. The "core principles" presented there were

- Manage information for all;
- Maintain individual motivation and demonstrate organizational commitment;
- Promote communication and public dialogue;
- Ensure fair and sound decision making and decisions; and
- Build and maintain trust.

The WateReuse Foundation (2004) summarized what it termed "best practices" to ensure that "well planned indirect potable reuse projects receive fair consideration in water supply decisions." It listed 25 such best practices, many of which overlap with others discussed here. Some of those considered to be the most critical are summarized below.

- Stakeholders will likely support a project if they understand that the project will improve their quality of life. Stakeholders must be able to perceive the value of the project. This is done through public education and collaboration.
- A water agency must take the leadership to clearly articulate the problem to stakeholders. For example, a continuing dialogue with the community about water supply and drought resistance should be held prior to identifying a solution. A water agency should help communities define the value of the project. Communication between the water agency and the stakeholders should be continuous and should start early in the process. A water agency should develop and maintain good relationships with key audiences such as elected officials, the media, the community, and other official decision makers.
- Once a problem has been defined and understood, all alternatives should be reviewed. The value of each solution is assessed in relation to other alternatives. Potential for conflicts are possible when a particular solution or project appears to have been forced on stakeholders. When conflicts do occur, the water agency should endeavor to understand the issues underlying conflict and opposition and deal with them constructively. Water agencies should be advocates for solving the problem, not advocates for a particular project.
- Water agencies should be cautious of any environmental justice issues that may arise. A project that is first implemented in neighborhoods

where community leaders reside may be perceived better than one where they do not.

- Establishing that a water agency is a trusted source of water quality is important. While the public generally credits the quality of water to its original source (i.e., a spring, a river, groundwater), the reality is that the safety of water is generally due more to the diligence of a water utility and its investments in testing and treatment. It is a high priority for the water agency to be perceived as the source of that quality.

It should be noted, of course, that many of these practices could be also usefully employed by opponents of potable reuse or related projects.

Forester (1999) provides a thorough discussion of participatory decision making processes that is quite consistent with these principles. The NRC (2005) also appointed a committee to study the state of knowledge on public participation in environmental decision making, and the committee's observations similarly stressed the importance of early involvement, open sharing of information, and solicitation of citizens' opinions in environmental and natural resource decisions.

Typically more concerns are raised regarding well recharge systems, partially due to association of these programs with disposal of wastes, versus managed underground storage of water intended for later recovery. Two examples of public involvement and public perception concerns that have arisen in specific MUS projects follow below.

Orange County, California

The Orange County Water District in Orange County, California, has implemented a successful public involvement program as part of its Groundwater Replenishment System, a managed underground storage project. In addition to community research and program evaluation activities, the public outreach effort included community presentations, appearances on local and public access cable television programs, distribution of materials to and through libraries and other public gathering places, a media relations program, and site and project tours (Wildermuth, 2001). District staff made an average of 120 presentations per year for seven years, to a wide array of civic groups, not only environmental, business, or other obviously interested organizations. The district's program of information and engagement is generally credited with the general public approval of the groundwater storage project, even though reclaimed wastewater is a significant source of the water being stored for later recovery, blending, and further treatment (Boxall, 2006).

Georgia and Florida

Public perception of a technology such as ASR can be influenced positively or negatively depending on the degree of open and well-communicated scientific facts and uncertainties.

In January 2001, a bill was before the Georgia General Assembly that would place a moratorium on injection of treated river water into the Floridan Aquifer System (FAS) through ASR. This moratorium gained the support of environmental advocates; one politician wanted to make the moratorium permanent and statewide. Another politician recognized that "not enough is known scientifically on whether the technology is safe or potentially harmful" and therefore was uncertain regarding whether "we need to close that door completely right now" (Florida Times Union, January 13, 2001). The bill was not signed into law by the governor (to update before publication).

A few months later in Florida, a different message was being communicated. Florida's ASR operations had existed for several years with few known problems. The recharged water was (and is) required to be treated to drinking water standards. However, legislation before Florida state lawmakers was being proposed to relax water quality standards (e.g., fecal coliforms) prior to recharge into the FAS under certain conditions. Concurrent with these legislative actions was the release of a report by the NRC (2001) that identified issues of uncertainty regarding the role of ASR in the $7.8 billion Everglades restoration plan. Moreover, the New York Times (April 13, 2001) reported that "the state is in the midst of a drought that is the worst in 50 years," and forecasts say that by 2020 without new sources Florida "would face a water deficit of as much as 30 percent." As such, the ASR issue was very high profile.

As the Florida bill successfully moved forward during the 2001 legislative session, newspaper articles described the bill as allowing "untreated," "polluted," or "tainted" water to be injected into the FAS. The bill was amended often to address environmental and scientific concerns as they arose; however, public opposition was building. The message from the legislature differed from that of environmental advocates, and scientists could not provide definitive answers. On April 24, 2001, an editorial in the St. Petersburg Times by a state environmental protection official addressed misperceptions regarding the bill and acknowledged that "opponents of the ASR plan have done a masterful job of offering sound bites that would ignite most who were hearing of the legislation for the first time. The problem is, many of these sound bites are false." After considerable debate among lawmakers and strong opposition by environmental advocacy groups, the bill was withdrawn. This occurred even though many believed the bill as amended in its final version provided conditions to ensure protection of Florida's groundwater resources.

In Georgia, uncertainty led an effort to remove ASR as a water-resource management option, while in Florida, this established technology experienced a perception "backslide" due to: (1) introduction of a legislative bill that would have benefited from additional input by scientific and technical experts, and (2) communication of a mixture of facts, uncertainty, and inconsistent, misleading, or misunderstood (thus poorly communicated) information. Although the bill rapidly evolved to address concerns, change in public perception outran its progress. Today, the pros and cons of ASR in Florida are widely understood (e.g., NRC, 2001) and have had the benefit of increased scientific study, open communication, and the support of a more informed citizenry.

FINANCIAL DRIVERS AND RELATED CONSIDERATIONS

Chapter 5 discusses many of the economic issues associated with MUS. Financial considerations are an important component in the development of MUS. Major financial considerations are capital costs and operating costs. Availability of grants, loans, and other subsidies and rate-paying schemes are also other financial considerations.

Capital costs include the cost of the land needed for recharge facilities, the cost of constructing recharge facilities, and the cost of surface water retention and conveyance facilities necessary to capture and move the water to recharge facilities. Operational and maintenance costs include cost of the water to be stored, cost of any additional treatment required, cost of acquiring the necessary easements and permits, and monitoring costs. For some systems using reclaimed water, monitoring costs may be a significant factor affecting the final cost of the stored water and the feasibility of the MUS project.

The large initial capital costs of such projects may be beyond what can be covered by tax increases, special assessments, or user fees. In such cases, water agencies tend to finance some of the capital costs through bonds, loans, and grants. Of these, bonds have been one of the most commonly employed methods of public finance (Howitt et al., 1999). In some cases, local agencies can utilize tax-exempt bonds as an effective approach to generate funding. With this approach, revenue from the project is used to support the debt. Debt servicing is usually the largest cost component of these projects. The incremental increase in user fees associated with project costs will depend on the cost of the project, the size of the user base, and other factors.

A major challenge to MUS is the ability of water providers to secure the financing necessary to develop a project where infrastructure is needed to bring surface water into a site that is feasible for recharge. In some cases, the distance between the location of available surface water and the recharge site may be large and there may be no existing infrastructure to convey the water. In other cases, the cost of land for recharge facilities is prohibitive. Small water providers may be limited in gaining access to such infrastructure and resources without the support of larger-scale water providers or without institutional coordination.

Individual states sometimes impose limitations on the powers that local governments have to raise funds to secure needed financing. Even where public entities recognize opportunities to possibly pool resources or coordinate funding, if the legal mechanisms are absent to facilitate this, then project implementation is less likely.

As noted in Chapter 5, it is possible for a project to be financially justified but not economically justified. It is also possible for a project to be economically justified but not financially feasible. Important factors that relate to financial feasibility include whether institutional opportunities are available to make a project feasible and whether the legal authority exists to support it. If institutional opportunities are not available or legal authority does not exist to support it, then project proponents may need to evaluate other institutional arrangements. Such arrangements may involve the creation of a new agency to sponsor the project, which may entail increased costs for the project.

Historically, surface storage projects have been heavily subsidized through agencies such as the Bureau of Reclamation and various state agencies, especially in the western United States. These subsidies can take the form of grants or low-interest loans for capital improvements, and operating subsidies based on the amount of water stored underground. Issues concerning subsidies have been addressed in Chapter 5.

Revenues from MUS projects are obtained through the sale of the stored water. Collecting revenues from water users is a critical financial consideration when planning for MUS. Particularly, it is not clear how MUS will affect water rates. Pyne (2005) identifies two issues: timing of when consumers pay for water stored and not yet recovered; and how costs can be distributed among users with very different demands. Some rate-paying arrangements are evolving as in Pasadena, California, where the city has to provide storage capacity for the Metropolitan Water District of Southern California (MWD) for $3.00 per year per acre foot payable upon recovery (Pyne, 2005).

One major incentive cited by ASR facility owners or operators for using this technology is as a means of maximizing the use of water treatment facilities. Shrier (2002) found that ASR facilities intended for potable uses typically treat water to primary and secondary drinking water standards prior to recharge; a few (27 percent of responding facilities) perform some additional pre-recharge treatment at the wellhead (e.g., pH adjustments) to improve injection operations and prevent geochemical interactions between the stored and native waters underground. Most ASR facilities perform no additional post-recovery treatment before introducing the recovered water into their water supplies. 42percent of the responding facilities perform minimal post-recovery treatment prior to recharge (e.g., pH adjustments, iron and manganese removal, filtration or turbidity reduction).

Thus, ASR enables facility owners and operators to shift the demand on treatment facilities to non-peak periods by treating the stored water to drinking water standards prior to recharge. The capacity of water treatment facilities is typically designed to meet peak treatment demands. Increasing non-peak use and decreasing peak use of water treatment facilities enable water providers to delay the need for capital investments for increased treatment capacity.

Another financial incentive for groundwater-dependent utilities is the way in which wholesale water agencies purchase contracts or rates are structured. For example, in Wildwood, New Jersey, retail water agencies pay for treated water from the wholesale water agency whether they use the water now, store it for later use, or do not use it at all. This rate environment creates an incentive to capture the water during periods of low demand and store it underground for use during periods of high demand. (Wildwood has highly seasonal demand as a coastal resort area with a much greater summer than winter population.)

Until 2005 the water stored underground in New Jersey had to be extracted within one year of storage or be claimed by the state. New Jersey has recently allowed studies on water banking for longer periods in the northeast part of the state. In an effort to restore seriously overdrafted regional aquifers the state allows only 85 percent of the water stored underground to be extracted and recovered in designated critical zones of the state. The remaining 15 percent of stored water reverts to the state.

To ensure reliable customers for the new 30 million-gallon-per-day water treatment and conveyance facilities, purchase contracts from the regional wholesale agencies in New Jersey American and South Jersey have been structured based on consistent use over 365 days each year. The contracts are essentially take-or-pay contracts. Since the water must be paid for regardless of whether it is used, the water purchasers have virtually no marginal cost for the water during low-demand periods. In some areas the demand rate is set at 90 percent of the full water rate, so the marginal cost is essentially 10 percent of the full cost for the water. Under these circumstances storing the water underground until it is needed the following summer has become an easy decision for Wildwood.

In Southern California, the water stored in the ground for MWD can be called under various constraints, but it is generally expected to be stored for several years and available under drought conditions. Such storage is being called upon along with surface storage during 2007 to cope with the severe shortfall in water available from imported water sources, the Colorado River, and the State Water Project. In addition to relatively new storage agreements, MWD has long relied on the availability of groundwater from basins that receive discounted replenishment water. The replenishment water program has involved no formal obligation to increase groundwater withdrawals during periods of need, but agencies such as OCWD have historically cooperated in increasing available groundwater during drought conditions affecting imported water supplies.

CONCLUSIONS AND RECOMMENDATIONS

Conclusion: The development of an MUS system from project conception to a mature, well functioning system is a complex, multistage operation requiring interdisciplinary knowledge of many aspects of science, technology, and institutional issues.

Recommendation: A comprehensive decision framework should be developed to assist in moving through the many stages of project development in an organized, rational way. Professionals from many fields, including chemists, geologists, hydrologists, microbiologists, engineers, economists, planners, and other social scientists should be involved in developing this framework.

Conclusion: Growing experience with MUS systems indicates that hydrogeological feasibility analysis including aquifer characterization is one of several important components in their development and implementation. The benefits of doing so include establishing the hydraulic capacity, recharge rates, residence times, and recoverable fraction of the introduced water—all of which help identify the optimum design and viability of the MUS system.

Some types of aquifers have matrix, hydrogeologic, and geochemical characteristics that are better suited to MUS systems than others. For example, the aquifer characteristics may dictate recharge, storage, and recovery methods. For an unconfined aquifer, source water can be recharged into the aquifer through recharge basins, vadose zone recharge wells, and deep recharge wells. Stored water can be recovered by production wells or ASR wells, or it can enhance baseflow to neighboring streams. For confined aquifers, however, source water can only be injected through deep recharge wells, including ASR wells. The stored water is usually recovered through ASR wells or downgradient production wells. As another example, water quality benefits are likely to be greater with alluvial systems compared to fractured or dual porosity systems.

Recommendation: Multiple factors should be assessed and monitored during design, pilot tests, and operations, including spatial and hydrogeological characterization of storage zones, temporal variation in quality and quantity of recharged, stored, and recovered water and factors that constrain sustainability of the MUS system, including hydrogeochemical, microbiological, and economic conditions. Uncertainty reduction is the ultimate goal.

Conclusion: An independent advisory panel can provide objective, third-party guidance and counsel regarding design, operation, maintenance, and monitoring strategies for an MUS project. An independent panel can increase public acceptance of and confidence in the system, if such trust is warranted. It can also be a catalyst for altering a plan if changes appear to be necessary.

Recommendation: Water agencies should highly consider the creation of an independent advisory panel or equivalent at an early stage of planning for an MUS system.

Conclusion: Relatively little research has been done to characterize the extent of vertical migration of fine-grained particles into the sediments beneath surface spreading facilities. Likewise, the science and technology of cleaning recharge basins is not well developed.

Recommendation: Research is recommended to develop new approaches to optimizing surface recharge, including assessing the extent of migration of fine-grained sediment into the subsurface, its impact on the long-term sustainability of surface recharge, and more efficient methods to clean recharge basins after clogging occurs

Conclusion: Successful MUS involves careful and thorough chemical and microbiological monitoring to document system performance and evaluate the reliability of the process. Each MUS project needs real-time monitoring of the quality of the waters being introduced into underground storage and of waters being extracted from storage for use.

Recommendation: Water quality monitoring programs should be designed on a case-by-case basis to assess water quality changes for elements, compounds, and microbes of concern, optimizing the potential for documenting any improvement in the quality of the source water and to collect samples representing any adverse water quality changes. A proactive monitoring plan is needed to respond to emerging contaminants and increase knowledge about potential risks.

Conclusion: New surrogates or indicators of pathogen and trace organic contaminant presence are needed for a variety of water quality parameters to increase the certainty of detecting potential water quality problems through monitoring. The categorization of chemicals and microorganisms into groups with similar fate and transport properties and similar behavior in treatment steps is one approach to streamline the list of potential contaminants to be monitored. It is unclear whether we can continue to rely on total coliform and *E. coli* indicator bacteria to characterize the microbial quality of water as the drinking water industry has done for decades. Such methodologies will improve the ability of MUS systems of a variety of sizes to engage in sound monitoring practices.

Recommendation: Research should be conducted to understand whether we can rely on monitoring surrogate or indicator parameters as a substitute for analysis of long lists of chemicals and microorganisms.

Conclusion: Surface spreading facilities sometimes require large amounts of land, particularly where large amounts of water are recharged or the geology is not ideal. Recharge well systems require less land, but may have as many different factors to consider in their placement. Optimization of recharge facility placement is important but not always well understood.

Recommendation: If there is some degree of freedom in site selection for recharge wells or basins, a location suitability assessment may be useful in site optimization. Factors such as ecological suitability, existing uses of the aquifer, groundwater quality, aquifer transmissivity, road density, land use and ownership, and access to power lines can be weighed in such an analysis.

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7
Managed Underground Storage in A Water Resource Systems Context

Las Vegas, Nevada, is not the only community in the United States that faces major uncertainties about its water supply. Nor are concerns about the future adequacy of water supplies confined to the arid and semiarid regions of the country. Even with increased water use efficiency and reduced per capita urban use, the prospects for continued growth of population and the economy will fuel increases in demand for municipal and industrial water. The demand for agricultural water is likely to increase as world demand for U.S. food and fiber grows with global population and as an increasing amount of irrigated acreage is put into production for the making of biofuels such as ethanol and biodiesel. Simultaneously, the continuing struggle to define and find adequate supplies of water to support environmental uses will contribute further to the growing national demand for water.

At the same time, there are reasons for believing that available water supplies may diminish. Groundwater overdraft tends to be the rule rather than the exception nationally. Ultimately, the inevitable decline of overdraft will lead to lower aggregate levels of extractions and a diminution of available supply. Threats to water quality will continue and some of those threats will materialize, leading to contamination episodes that may render some accustomed supplies unfit for use at least on a temporary basis. Finally, current and prospective global climate change threatens to alter both the magnitude water available in some regions and the timing of water availability. The result is that our water resource systems will be characterized by growing demands and static or shrinking supplies.

Indeed, such circumstances characterize the global water situation as well as that in the United States (Jury and Vaux, 2005). Globally, demands are increasing as population grows and an increasing numbers of countries no longer have sufficient water resources to provide the water services, sanitation, and food and fiber needs of their populations. The numbers of countries in these circumstances will grow in the next two decades.

Groundwater overdraft is pervasive worldwide and even more alarming than it is in the United States. Thus, for example, India and China are today feeding 400 million people with crops irrigated with unsustainable overdraft. It is not at all obvious where the water to feed these people will be found once the aquifers in question are economically exhausted (Jury and Vaux, 2005). In the United States, total withdrawals of freshwater from 66 major aquifers were estimated at 93.3 million acre-feet (83,300 million gallons per day [Mgal/d]) for the year 2000 (Maupin and Barber, 2005). Many of these aquifers are receiving only small amounts of recharge, and considerable storage space has been created through these long-term withdrawals. In particular, the greatest available storage for development of managed underground storage (MUS) systems may be in unconsolidated and semiconsolidated sand and gravel aquifers. Thus, in the appropriate circumstances, managed underground storage will offer communities and regions throughout the world an opportunity to address problems of overdraft as well as another tool that can be integrated into a balanced system for managing water scarcity.

The growing scarcity of water will require that water be managed more carefully and that it be used more intensively. More intensive use implies that the productivity of water in existing uses will have to be increased and also that waters that currently are not used, or are underutilized, must be the object of more intensive exploitation. Examples include flood flows (Boxes 5-4 and 5-5), urban stormwater (the stormwater-to-drinking water project in Salisbury, Australia described in Chapter 6) and reclaimed wastewater (numerous examples throughout the report).. Similarly, water will have to be managed in an integrated unified way that acknowledges explicitly the interrelatedness of the hydrologic cycle and the interrelatedness of water and other natural resources. This represents a departure from the way water resources have been developed and managed in the United States.

Until the late to mid–twentieth century the primary means of responding to water scarcity was to build surface water storage and conveyance projects. Surface water storage ultimately fell from favor because: (1) the low-cost sites were soon all developed, leaving only opportunities that were considerably more expensive; (2) the costs of constructing civil works projects grew faster than other costs in the economy; (3) the competition for public funds became keener, making it more difficult to secure the financing necessary to construct large surface water facilities; and (4) the environmental damages and social impacts associated with the construction and operation of surface water storage facilities became fully manifest at a time when public environmental awareness was growing.

There followed a period in which the emphasis shifted away from surface storage toward programs of conservation and more intensive management of water supplies. Improved techniques for managing water on-farm were devised and disseminated. Improved technology, including closed conduit irrigation systems and water saving appliances, were developed and adopted relatively widely. The public became more aware of water and more aware of behaviors that economize on water use. Water transfers began to be accepted. Transfers included the trading of water rights and entitlements, the purchase of water in spot markets, and the development of contingent markets for water. Such transfers have the capacity to reallocate water away from relatively low-valued uses to relatively higher-valued use, thereby ensuring that the productivity of water is optimized. They have the added advantage of being voluntary so that no one is coerced into participating in a water transfer.

There has also been a returned appreciation for the importance of some form of storage as a means of capturing and holding water that is available only during wet seasons or years and of keeping that water in a location where it can be accessed readily to meet demands, whether for municipal, industrial, agricultural, or environmental and habitat purposes. Several methods of water storage have recently been explored and utilized that do not involve use of larger, on-stream reservoirs, including tanks and towers, former mines, and gravel pits; each of these has its own challenges and benefits.

With these various water supply storage and management tools available, it has been possible to address many water scarcity issues by relying predominantly on a single strategy, such as surface water storage, or a single subset of strategies like conservation measures and transfers.

However, in the future no single strategy, or even a small subset of strategies, is likely to suffice. Even the recycling of water has its limits due to consumptive use on each cycle. Rather, what will be needed is an integrated strategy in which all measures for managing water scarcity are considered and, if appropriate, employed in a balanced, systematic fashion. In an earlier report, a committee of the National Research Council (NRC, 2004) called for water to be viewed and managed in a broad systems context The methods and techniques for managing scarcity will also have to be cast in a broad systems context so that water resources can be managed in an integrated fashion that acknowledges the interrelatedness of the hydrologic cycle and among natural resources. Managed underground storage will have to be part of this broad balanced strategy.

Although much can be accomplished through programs of water conservation, careful management of water, and the utilization of markets to accomplish water transfers, additional water storage will be required as the population and economy of the nation continue to grow. Managed underground storage has become attractive because it offers many of the benefits of surface water storage, often at less expense and without the environmental damages associated with surface water storage projects. Thus, for example, MUS can be employed to "firm up" water supplies that are highly variable across seasons and years. In a related way, managed underground storage can be utilized to provide drought protection and protection against the failure of surface infrastructure systems that are vulnerable to earthquakes and other natural hazards. It can also be used to improve the financial and operational efficiency of water production facilities, such as desalination and water purification plants, allowing these facilities to operate at relatively steady levels of output despite seasonal variability in water demands. In a word, storage increases the flexibility with which water can be managed.

Beyond these benefits, which can be attributed to virtually any type of water storage, managed underground storage has the added benefit that it can be used to attenuate or eliminate groundwater overdraft, and such systems can also provide conveyance *in lieu* of expensive surface water conveyance systems. Thus, there are a number of instances where managed underground storage projects have been used to meet growing demands from newly developed areas in order to avoid the costs of expensive surface conveyance systems. In addition, given the magnitude of annual groundwater overdraft in the United States, as well as, its cumulative magnitude, there is already an enormous amount of storage capacity underground and that capacity is growing daily. As a consequence, MUS has a very important potential role in addressing the intensifying water scarcity of the future.

Collective experience with MUS systems is substantial. A significant number of these systems are decades old, and experience indicates that many of them perform consistently and well over the longer term. However, managed underground storage is not a panacea. It is likely to be costly, although it is increasingly the least-cost alternative, and cannot by itself resolve all of our water scarcity problems. By reducing stormflow or return flows from wastewater treatment plants to streams, it can decrease water availability for people or ecosystems downstream. It can, however, play a very important role in balanced programs to manage water scarcity.

In anticipating, planning for, and developing MUS projects, it will be vital to bear in mind the need to view and manage water in a broad systems context. This can be greatly facilitated by the existence of regional water districts, authorities, and agreements of various kinds. For example, since 1972 Florida has been divided into five water management districts, which roughly correspond to surface watersheds; the South Florida Water Management District coordinates water supply, ecosystem restoration, and coastal and terrestrial water quality on a large scale. Other efforts are much newer. For example, the South Metro Water Supply Authority (Denver) was formed in 2004. It is composed of 13 water providers that have created a single master plan to foster long-term reliable water supplies through water acquisition and infrastructure.

An extreme example of a regional water management approach is an agreement by which Arizona stores Colorado River water in an aquifer on behalf of Nevada. When Nevada needs to recover the water, it withdraws a quantity of Arizona's Colorado River water directly from Lake Mead, while Arizona withdraws the equivalent amount of water from the aquifer.

Integrated programs of water management will differ from place to place in the balance of measures ultimately selected and in the water management schemes employed. This, of course, is a straightforward consequence of the fact that most potable water is supplied at fairly local scales and involves decisions by municipal or county politicians responsible to local constituencies. However, the complex water management challenges described throughout this report generally require a broad systems approach in conceiving, designing, building, and operating MUS projects. Six elements of such an approach are summed up in the following paragraphs.

First, it is imperative that the connections between ground- and surface water be acknowledged and recognized in conceiving and designing such project. Ground- and surface waters are frequently interconnected, and alterations in the state of groundwater, for example, can have unintended consequences for interconnected surface waters. As discussed in earlier chapters, groundwater pumping can lower or even eliminate baseflow in streams that support fisheries, agriculture, or other uses. Surface water diversions to supply water for recharge basins may have a similar effect. Such interconnections need to be understood and acknowledged in project planning and operations.

Second, the interdependent nature of ground- and surface water quality also needs to be acknowledged. Waste sinks—land, air, and water—cannot be managed in isolation from one another. In part, this is because a high qualitative standard in one sink implies a low quality standard in either or both of the other sinks. Similarly, the quantitative and qualitative status of surface water has implications for the quantitative and qualitative status of groundwater and vice versa. There are two mechanisms of connection: (1) degradation of surface water quality can ultimately lead to degradations in groundwater quality and vice versa; (2) regulations that call (for example) for a high standard for groundwater and a low or no standard for surface water inherently ignore the interconnectivity and hydrological interrelatedness of water resources. The circumstances in which it makes sense to lower the quality of water in one place (e.g., surface water) in order to protect it in others (e.g., groundwater) are few. At the same time it is important to recognize the differences in waste assimilative capacity of surface and groundwaters in designing appropriate regulations.

Third, with the greater use of MUS projected for surface water, including stormwater, the integration of water supply and water pollution control will be even more crucial in the future. Rather than depending primarily on technology for water treatment, controlling contamination of streams from combined sewer overflows, failing septic systems, and agricultural and urban runoff will be an important part of the solution (NRC, 2000, 2005). Since many of the regions considering MUS are located along or near coastlines, which are characterized by both intensive recreational activity and highly sensitive ecosystems, the quality and quantity of water that reaches the coast is often of keen interest to water and natural resource managers (NRC, 1993). Integration of water supply and storage with stormwater runoff, pollution control, and coastal management is therefore a strategy that provides benefits across a broad sector of society.

Fourth, there is a need for additional specific research efforts in order to facilitate the development and implementation of MUS schemes. Some of this research is necessarily local and focused on a specific water system and its alternatives. However, this research should complement regional and national studies that, in turn, form part of an integrated national program of water resources research (NRC, 2004). The research recommended in this report does not stand in isolation from other areas of water research and is not necessarily of higher priority than all other water research needs. A reinvigorated program and its priorities need to be developed by a national partnership that includes federal, state and local interests as well as representative stakeholders (NRC, 2004).

Fifth, there is a need for data and monitoring in connection with the development and operation of MUS projects. This need should be viewed from the perspective of a larger national need to reinvest in monitoring, data acquisition, and data retrieval for water resources. Specifically, the trend of disinvestment in water resources monitoring, data acquisition, and retrieval needs to be reversed soon if the nation is to address successfully, and at reasonable costs, its mounting water resources problems. The need for data on generalized aquifer characterization and specific aquifer properties should be viewed as one element in a collaborative national program of monitoring and data collection for water resources.

Sixth, there is a compelling need to devise appropriate institutions for the management of underground storage systems as well as the broad integrated systems that will ultimately be required to manage water scarcity. Existing laws and regulations are often inconsistent. Frequently, it is difficult to make them both effective and flexible. Management responsibilities are often fragmented among agencies, with the result that programs are uncoordinated, transactions costs are higher than they need to be, and integrated approaches to water problems are difficult to develop. Agency missions and programs are sometimes too narrowly conceived as with single-purpose agencies and programs whose mandates effectively prohibit them from viewing water problems in a broad systems context. Many existing institutional arrangements embody poorly conceived incentives that have led to unintended consequences. Also, many existing water institutions were designed in different eras for different purposes and are now ill-suited to address contemporary and future problems. Despite the compelling need for innovative institutions, support for research on institutional topics—research that may be the basis for institutional innovations—has fallen to near zero in recent decades. The need for modern institutions capable of managing underground storage, as well as water management systems in general, both efficiently and effectively is clear and urgent (NRC, 2001, 2004).

Overall, albeit focused on issues surrounding managed underground storage, this report has highlighted the complexity of modern water management, especially in areas facing population growth, increasing competition for water for energy and the environment, earlier seasonal snowmelt, and other climate change issues. It has underlined how the interconnectedness of groundwater and surface water has opened new opportunities for conjunctive water management and identified potential risks to human and environmental health. It also has underscored some of the challenges to finding creative solutions in a legal and regulatory environment that was not created to facilitate such solutions. While managed underground storage is just a part of the answer to society's water management challenges, it is hoped that cities, states, and the nation will devote the necessary resources to learning how important a role MUS can play on a national scale.

CONCLUSIONS AND RECOMENDATIONS

Conclusion: Although failures have occurred and the potential for contaminating groundwater is a considerable risk, most MUS systems have successfully achieved their stated purposes. In fact, there are MUS systems that have functioned without major problems for decades. However, increasing efforts to use karst and fractured aquifers for storage will increase the potential for failures. Chemical reactivity of the aquifer in the former case, and uncertainty over flow paths in either case are much greater and the treatment potential is lower compared to alluvial aquifers. Learning from past positive and negative performance will help guide development of the many new MUS systems that are under consideration.

Recommendation: Given the growing complexity of the nation's water management challenges, and the generally successful track record of managed underground storage in a variety of forms and environments, MUS should be seriously considered as a tool in a water manager's arsenal.

Conclusion: In the future, multiple strategies are likely to be needed to manage water supplies and meet demands for water in the face of scarcity. Various water conservation and management strategies, including transfers and water recycling, can be used to stretch available water supplies. However, each of these has its limits. The use of water storage facilities remains an essential component of water management, particularly in areas where water availability varies greatly over seasons or years, such as the arid Southwest. Integrated strategies will be needed in which all measures for improving water quality and managing water scarcity are considered and, if appropriate, employed in a balanced, systematic fashion. Seasonal to multiyear storage of water will often be a necessary component of such strategies.

Recommendation: In anticipating, planning for, and developing MUS projects, water managers should consider the role and merits of MUS in conjunction with other water management strategies.

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Acronyms and Glossary

Abiotic	Refers to chemical transformations that occur without the aid of microorganisms.
Acre-foot	A traditional measure of water applied, used in the United States. The volume of water required to cover 1 acre of land to a depth of 1 foot. Equal to 1.23 ML or 1,230 m^3..
Adsorption	The adherence of ions or molecules in solution to the surface of solids.
Advanced wastewater treatment	Any physical, chemical or biological treatment process used to accomplish a degree of treatment greater than that achieved by secondary treatment.
Advection	The process whereby solutes are transported by the bulk mass of flowing fluid.
AL	Elastic wave propagation log.
Anisotropy	The condition under which an aquifer property varies with the direction of measurement. For example "If the hydraulic conductivity, K, varies with the direction of measurement at a point in a geologic formation, the formation is anisotropic at that point." (Freeze and Cherry, p. 32)
Anoxic	Describes an environment without oxygen.
Aquifer	A formation, group of formations, or part of a formation that contains sufficient saturated permeable material to yield significant quantifies of water to wells and springs.
Aquifer storage and recovery (ASR)	Injection of water into a well for storage and recovery from the same well.

Aquifer storage transfer and recovery (ASTR) Injection of water into a well for storage and Recovery from a different well, generally to provide additional water treatment.

Artificial recharge (AR) Intentional banking and treatment of water in aquifers.

Artificial Recharge and Recovery (ARR) Recharge to and recovery of water from an aquifer, that is, both artificial recharge of the aquifer and recovery of the water for subsequent use.

AT Acoustic televiewer.

Augmentation pond Water body designed to supply water to river systems at defined rates during particular times.

Bank filtration Extraction of groundwater from a well or caisson near or under a river or lake to induce infiltration from the surface water body, thereby improving and making more consistent the quality of water recovered.

Base flow That portion of a stream's flow derived from ground water (as opposed to surface runoff and interflow).

Basin (1) *Hydrology*: The area drained by a river and its tributaries. (2) *Irrigation*: A level plot or field, surrounded by dikes, which may be flood irrigated. (3) *Runoff control*: A catchment constructed to contain and slow runoff to permit the settling and collection of soil material transported by overland and rill runoff flows

Basins and watersheds Areas of drainage in which all collected water ultimately drains through a single exit point. Basins differ from watersheds only in the perception of their size: basins are usually considered to be much larger, composed of many watersheds. Within a watershed or basin, water moves both on and below the surface. Topographic "highs" prevent surface water from crossing from one watershed (aquifer) to another.

Biodegradation	The biologically mediated conversion of a compound to simpler products.
Bioremediation	Exploiting the metabolic activity of microorganisms to transform or destroy contaminants.
Carbonate	A rock formed primarily from carbonate minerals, such as limestone and dolomite.
CERP	Comprehensive Everglades Restoration Program.
Chlorinated solvent	One that contains at least one chlorine atom. Typically, these compounds are used to dissolve substances that do not dissolve easily in water. Because they are used for a wide variety of purposes—from manufacturing, to degreasing, to dry cleaning—chlorinated solvents are common groundwater contaminants.
Colloid	A particle that has a diameter in the range of 10^{-8} to 10^{-5}m. The small size of colloids tends to keep them in suspension for long periods.
Complexation	A reaction in which a metal ion and one or more anionic ligands chemically bond. Complexes often prevent the precipitation of metals.
Confined aquifer	An aquifer bounded above and below by units of distinctly lower hydraulic conductivity in which the pore water pressure is greater than atmospheric pressure. An **unconfined** aquifer is not bounded above and is the uppermost aquifer.
Conjunctive use	Combining the use of both surface and groundwater to minimize the undesirable physical, environmental, and economic effects of each.
Consumptive Use	Use of water that renders it no longer available because it has been evaporated, transpired by plants, incorporated into products or crops, consumed by people or livestock, or otherwise removed from water supplies.

CSMAT Controlled source audio-frequency magneto-telluric

Cumulative recovery efficiency Ratio of the cumulative volume of freshwater injected minus the volume of unrecovered fresh water divided by the cumulative volume fresh water injected.

CZ Confinement zone.

Darcy's Law A formula used to describe fluid flow in the subsurface. The law states that the velocity of flow through a porous medium is directly proportional to the hydraulic gradient (assuming that the flow is laminar and inertial forces can be neglected).

Denitrification The conversion of nitrate to nitrogen gas by microorganisms. Denitrification can be an important process in the subsurface, because when oxygen is absent, denitrifying bacteria can use nitrate to degrade hazardous compounds in the same way that they would ordinarily use oxygen.

Density The mass per unit volume of a substance.

Desorption The release of sorbed molecules from solid into solution (the reverse of sorption).

Diffusion Contaminant movement caused by the random motion of molecules. Contaminants diffuse from areas of high concentration to areas of low concentration.

Disinfection by-products A range of organic and inorganic products resulting from the reaction of disinfecting oxidants with natural aquatic organic material reductants in water systems. The number and nature of all products are not precisely known at present, and vary with type of disinfectant employed. Some of the chlorination by-products are mutagenic and some are suspected animal carcinogens.

Dispersion The spreading and mixing of chemical constituents in groundwater. Dispersion is caused by diffusion and mixing due to microscopic variations

	in velocities within and between pores as well as by macroscopic velocity variations among zones of differing hydraulic conductivity.
Dissolution	The process by which solid- or nonaqueous-phase liquid components of a contaminant dissolve in infiltration water and form a groundwater contaminant plume. The duration of remediation measures (either cleanup or long-term containment) is determined by the rate of dissolution that can be achieved in the field and the mass of soluble contaminants.
DOC	Dissolved organic carbon.
Drawdown	Lowering of the water table or potentiometric surface as a result of pumping.
Dry well	Synonymous with *vadose zone well*.
Enteric viruses	Members of a large group of viruses characterized by the fact that they replicate in the intestinal tract and are therefore present in fecal material.
EPMA	Electron probe microanalysis.
ERT	Earth resistivity tomography.
Evapotranspiration	The sum of evaporation and transpiration from a unit land area. Also see **consumptive use**.
Fractured media	Large subsurface rocks or clay formations that are mostly solid but contain cracks that can transmit or store water.
GGL-D	Gamma-gamma log.
GPR	Ground-penetrating radar.
GR	Gamma-ray log.
Groundwater	That part of the subsurface water that is in the saturated zone.

Ground water overdraft (or mining) The withdrawal of groundwater through wells, resulting in a lowering of the ground water table at a rate faster than the rate at which the ground water table can be recharged.

GWR Groundwater replenishment.

Halogenated compound A compound in which one or more hydrogen atoms have been replaced by a halogen atom, such as fluorine, chlorine, or bromine. Examples include chlorinated solvents (such as 1,1,1-trichloroethane, trichloroethylene, and tetra-chloroethylene), which have been widely used in cleaning and degreasing operations in some fu-migant pesticides. Many halogenated compounds are DNAPLs.

Head The pressure of a fluid on a given area, at a given point caused by the height of the fluid surface above the point. Also, water-level elevation in a well, or elevation to which the water of a flowing artesian well will rise in a pipe extended high enough to stop the flow.

Heterogeneity Pertaining to an aquifer, variation in the value of one (or more) measurable properties in space. A synonym is nonuniform.

Homogeneity Refers to subsurface media that are relatively uniform.

Humic substance A macromolecular organic substance formed from the decomposition of plant or animal mate-rial.

Hydraulic barrier A barrier to flow caused by system hydraulics, such as a line of ground water discharge caused by extraction wells.

Hydraulic conductivity (*K*) The coefficient of proportionality between the flow rate (specific discharge) of water through a permeable medium in response to a hydraulic gradient. The density and kinematic viscosity of the water affect the hydraulic conductivity. It has dimensions of L/T. K is a function of both the permeable medium and the fluid moving through it. It is related to the intrinsic permeability (k):

$K = k(\rho g/\mu)$,

where ρ represents the fluid density and μ represents the dynamic viscosity.

Hydraulic gradient Difference in hydraulic head between two points divided by the distance between the points.

HFO Hydrous ferrous oxide.

Hydrophilic "Water loving"; refers to compounds that are highly water soluble.

Hydrophobic "Water fearing"; refers to substances that are relatively insoluble in water.

Igneous rock A rock that solidified from molten material. "Igneous" is one of the three categories (igneous, metamorphic, and sedimentary) into which all rocks are divided.

Infiltration The flow of water downward from the land surface into and through the upper soil layers.

Infiltration Basin Synonymous with recharge basin.

Infiltration rate Generally, the rate at which a soil under specified conditions can absorb falling rain or melting snow; in recharge, the rate at which water drains into the ground when a recharge basin is flooded, expressed as of water per unit time.

Injection well Well used for emplacing fluids into the subsurface.

Intrinsic permeability A measure of the relative ease with which a porous medium can transmit a liquid under a potential gradient. Intrinsic permeability is a property of the medium and is dependent on the shape and size of the openings through which the liquid moves.

Ion A molecule that has a positive or negative electric charge.

Ion exchange The exchange of ions between a solution and a solid while maintaining charge balance. Through ion exchange, charged molecules that are naturally part of the subsurface soil may be replaced by contaminant molecules.

Irrigation The application of water to soil for crop production or for turf, shrubbery, or wildlife food and habitat. Intended to provide water requirements of plants not satisfied by rainfall.

Leakance The ratio of vertical hydraulic conductivity (K_v) to the thickness of the confining unit or aquitard.

Lithology A description of the rocks beneath the ground at a site.

LL Conductively focused-current logs.

Managed (or management of) Aquifer Recharge (MAR) Intentional banking and treatment of water in aquifers (synonymous with AR). MUS may be considered a subset within MAR.

Maximum contaminant level (MCL) The maximum amount of a compound allowed in drinking water under the Safe Drinking Water Act. MCLs are set by considering both health effects of the compound and technical feasibility of removing the compound from the water supply.

Maximum contaminant levels goal (MCLG) Nonenforceable health goal established under the Safe Drinking Water Act intended to protect against known and anticipated adverse human health effects with an adequate margin of safety. Technical feasibility is not considered in setting MCLGs.

Method detection limit The constituent concentration that, when processed through a complete method, produces a signal with a 99 percent probability that it is different from a blank.

Metamorphic rock — A rock created from preexisting rocks in response to changes in temperature, pressure, shearing stress, or chemical environment.

MLL — Micro-focused logs.

Monitoring well — A tube or pipe, open to the atmosphere at the top and to water at the bottom, used for taking groundwater samples.

MUS — Managed Underground Storage.

NGW — Native groundwater.

NNL — Neutron log.

Numerical model — A model whose solution must be approximated by varying the values of controlling parameters and using computers to solve approximate forms of the model's governing equations.

Oxidation reaction — The transfer of electrons away from one compound to another.. Oxidation reactions are important in the destruction of contaminants. They may occur spontaneously when the appropriate chemicals are mixed, or they may be catalyzed by microorganisms. For example, when microbes degrade organic compounds, they may transfer electrons away from the compound, converting the compound to carbon dioxide and deriving energy from the electron transfer process.

Pathogen — A disease-causing microorganism.

Permeability — The coefficient of proportionality between the flow rate (specific discharge) of a fluid through a permeable medium in response to the hydraulic gradient (driving force); k is a characteristic solely of the medium. The dimensions of k are L^2. The relation to hydraulic conductivity is given in **Hydraulic conductivity**.

Phreatophyte	A deep-rooted plant that obtains its water from the water table or the layer of soil just above it.
Plume	A zone containing predominantly dissolved contaminants and sorbed contaminants in equilibrium with the dissolved contaminants. A plume usually will originate from the contaminant source areas and extend downgradient for some distance, depending on site hydrogeologic and chemical conditions.
Polychlorinated biphenyl (PCB)	A type of contaminant built from two benzene rings and chlorine atoms. PCBs are very stable, resisting both chemical and biological degradation, and are toxic to many species. At one time, they were used commonly in electrical transformers as heat insulators.
Polycyclic aromatic hydrocarbon (PAH)	A compound built from two or more benzene rings. Sources of PAHs include fossil fuels and incomplete combustion of organic matter (in auto engines, incinerators, and even forest fires).
Pore	A small space between the grains of sand, soil, or rock in the subsurface. Groundwater is stored and transmitted in pores.
Porosity	The ratio of the volume of void spaces (V_v) contained within a volume of rock, sediment, or soil, to the total volume V_t (rock, sediment or soil particle volume + void space volume) (porosity = V_v/V_t). The *effective porosity* represents voids spaces through which water or other fluids flow in a rock or sediment. It excludes isolated or dead-end pores and the volume within pores occupied by water adsorbed on minerals. *Primary porosity* is the space between grains created when a rock or sediment was formed. Secondary porosity is caused by fracture or weathering in a rock or sediment after it has been formed.
Porous medium	A subsurface zone composed of small rocks or sand particles with pores that can transmit or store water.

Potable reuse, direct	Occurs when there is a piped connection of water reclaimed from wastewater to a potable water supply distribution system or a water treatment plant.
Potable reuse, indirect	*Planned* indirect potable reuse occurs when wastewater effluent is discharged to a water source with the intent of subsequently reusing the water rather than as a means of disposal. *Unplanned* indirect potable reuse occurs when a water supply is withdrawn for potable purposes from a natural surface or underground water source that is fed in part by the discharge of a wastewater effluent. The wastewater effluent is discharged to the water source as a means of disposal and subsequent reuse of the effluent is a byproduct of the disposal plan.
Potable water	Water that has been treated to be or is naturally suitable for drinking.
Potentiometric surface	The height of rise of the water due to hydrostatic pressure when the constraint of the confining layer is removed. Sometimes referred to as the piezometric surface.
Prior Appropriation	A concept in water law under which a right is determined by such a procedure as having the earliest priority date.
QA/QC	Quality assurance/quality control.
ORE	Operational recovery efficiency.
Recharge area (groundwater)	An area in which water infiltrates the ground and reaches the zone of saturation.
Recharge basin (or pond)	A surface facility, often a large pond, used to increase the infiltration of surface water into a groundwater basin. Basins require the presence of permeable soils or sediments at or near the land surface and an unconfined aquifer beneath.

Recharge	The replenishment of water beneath the earth's surface, usually through percolation through soils or connection to surface water bodies.
Recharge Well	A well used to recharge water directly to either a confined or an unconfined aquifer.
Reclaimed water	Wastewater made fit for reuse for potable or nonpotable purposes.
Redox potential	The distribution of oxidized and reduced species in a solution at equilibrium. Redox potential is important for predicting the likelihood that metals will precipitate from ground water upon pumping, for estimating the capacity of microorganisms
Reduction reaction	The transfer of electrons to one compound from another (also see **Oxidation reaction**). Oxidation-reduction reactions are important in the destruction of contaminants. They may occur spontaneously, when the appropriate chemicals are combined, or they may be catalyzed by microorganisms. For example, when microbes degrade organic compounds, they may transfer electrons from the compound to oxygen, converting the oxygen to water.
Residence time	The average amount of time a fluid spends during transport through a unit volume of subsurface or a laboratory vessel.
Retardation	The movement of a solute through a geologic medium at a velocity lower than that of the groundwater. Retardation is caused by sorption and other phenomena that separate a fraction of the solute mass from the bulk groundwater.
Reverse osmosis	A highly efficient removal process for inorganic ions, salts, some organic compounds, and in some designs, microbiological contaminants. Reverse osmosis resembles the membrane filtration process in that it involves the application of a high feed water pressure to force water through semipermeable membrane. In osmotic processes,

water spontaneously passes through semipermeable membrane from a dilute solution to a concentrated solution in order to equilibrate concentrations. Reverse osmosis is produced by exerting enough pressure on a concentrated solution to reverse this flow and push the water from the concentrated solution to the more dilute one. The result is clear permeate water and a brackish reject concentrate.

Reynolds number Expressed as follows: $R = \rho v d/\mu$
Where ρ is the density of water (mass/volume), v is the specific discharge (length/time), d is a representative grain diameter for the porous media (often taken as the 30% passing size from a grain size analysis using sieves - units of length), and μ is the dynamic viscosity of the water (mass/(length x time).

Runoff That part of the precipitation that moves from the land to surface water bodies.

RW recharged water.

Safe Drinking Water Act (SDWA) The law, passed in 1974, that required the setting of standards to protect the public from exposure to contaminants in drinking water.

Salinization To become impregnated with salt; concentration of dissolved salts in water or soil water. An environmental impact of irrigation that can be managed but not eliminated.

Saturated zone That part of the earth's crust beneath the regional water table in which all voids, large and small, are filled with water under pressure greater than atmospheric.

Secondary porosity The porosity developed in a rock formation after its deposition or emplacement, either through natural processes of dissolution or stress distortion, or artificially through acidization or the mechanical injection of coarse sand

Sedimentary rock A rock created from the consolidation of loose sediment that has accumulated in layers.

Soil aquifer treatment (SAT) Treated sewage effluent, known as reclaimed water, is intermittently infiltrated through infiltration ponds to facilitate nutrient and pathogen removal during passage through the unsaturated zone for recovery by wells after residence in the aquifer.

Sorption A process that removes solutes from the fluid phase and concentrates them on the solid phase of a medium.

SP Spontaneous potential.

Specific capacity An expression of the productivity of a well. Obtained by dividing the rate of discharge of water from the well by the drawdown of the water level in the well. It has dimensions of $L^3/T\!-\!L$. It should be described on the basis of the number of hours of pumping prior to the time the drawdown measurement is made.

Specific storage The volume of water that a unit volume of porous medium releases from storage under a unit decline in hydraulic head (Freeze and Cherry) (while it still remains fully saturated). It has dimensions of inverse length, $[L^{-1}]$. It describes storage in confined aquifers.

Specific yield The term used to describe storage in unconfined aquifers. 'It is defined as the volume of water that an unconfined aquifer releases from storage per unit surface area of aquifer per unit decline in the water table' (Freeze and Cherry, p. 61).

Spreading basin Synonymous with **Recharge basin**.

Storativity The product of specific storage and aquifer thickness, defines the volume of water released from storage per unit decline in hydraulic head in the aquifer, per unit surface area of the aquifer

Stormwater runoff	Water resulting from precipitation which either infiltrates into the ground, impounds/puddles, or runs freely from the surface, or is captured by storm drainage, a combined sewer, and to a limited degree, by sanitary sewer facilities.
Sulfate reduction	The conversion of sulfate to hydrogen sulfide by microorganisms. Because they can degrade hazardous compounds without using oxygen, sulfate-reducing bacteria can be important players in the subsurface, where the oxygen supply is often limited.
Surface spreading	Recharging water at the surface through recharge basins, ponds, pits, trenches, constructed wetlands, or other systems.
Surface tension	The tension at the surface between a liquid and its own vapor.
Surficial aquifer	An aquifer that is near the earth's surface, in the most recent of geologic deposits.
SW	Source water.
TDEM	Time-domain electromagnetic.
TDS	Total dissolved solids.
Tertiary treatment	The treatment of wastewater beyond the secondary or biological stage. The term normally implies the removal of nutrients, such as phosphorus and nitrogen, and of a high percentage of suspended solids. It is now commonly replaced by the term “advanced waste treatment.”
TM	Temperature log.
TOC	Total organic carbon.
Transmissivity	The rate at which water is transmitted through a unit width of an aquifer under a unit hydraulic gradient. In a confined aquifer, it is equal to the product of the hydraulic conductivity and the aq-

	uifer thickness. It is a function of properties of the liquid, the porous media, as well as the permeability and thickness of the aquifer. It provides a measure of capability of the entire thickness of an aquifer to transmit water.
TSV	Target storage volume.
TW	Transitional water.
UIC	Underground Injection Control.
Unconfined aquifer	See **confined aquifer**.
Underground storage and recovery (USR)	Similar to MUS; any type of project whose purpose is the artificial recharge, underground storage, and recovery of project water.
Unsaturated zone	The zone between the land surface and the regional water table. Generally, water in this zone is under less than atmospheric pressure, and some of the voids may contain air or other gases at atmospheric pressure. Beneath flooded areas or in perched water bodies the water pressure locally may be greater than atmospheric.
Vadose zone	See **unsaturated zone**.
Vadose zone well	A well constructed in the interval between the land surface and the top of the static water level and designed to optimize infiltration of water.
Volatile organic compound (VOC)	An organic chemical that volatilizes (evaporates) relatively easily when exposed to air.
Wastewater	Water that carries waste from homes, businesses, and industries; a mixture of water and dissolved or suspended solids.
Water quality	The chemical, physical, and biological condition of water related to a beneficial use.
Water resource	The supply of ground- and surface water in a given area.

Water table	The "top" of the subsurface zone that is saturated with groundwater. More precisely, it is the surface in an aquifer at which pore water pressure is equal to atmospheric pressure.
Water Withdrawal	Water removed from ground or surface water sources for use.
Watershed	A geographic region (area of land) within which precipitation drains into a particular river, drainage system or body of water that has one specific delivery point.
Water-table aquifer	An aquifer in which the water table forms the upper boundary.
WTP	Wastewater Treatment Plant.

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Appendixes

Appendix A
Physical, Chemical, and Microbiological Constituents of MUS Waters

The purpose of this appendix is to expand on some of the brief descriptions of constituents that may be found in either recharge or discharge waters of managed underground storage (MUS) systems. These include basic physicochemical parameters, followed by inorganic and organic species, and finally microbes.

PHYSICAL CHARACTERISTICS

The first impressions of water quality are often based on visual observations. Water is expected to be free of particles (turbidity), color, and odor. Additional important physical characteristics of MUS waters include dissolved oxygen, pH, oxidation-reduction potential (Eh), specific conductance, and temperature.

Turbidity

Suspended particles impede the passage of light through water by scattering and absorbing the light rays. This interference of light passage is called turbidity. Waters with greater turbidity will experience increased clogging of filters and increased head loss development during infiltration. The particles contributing to turbidity can also harbor pathogens and enhance their survival in the presence of a disinfectant.

Color

Color in a water is usually the result of an elevated organic content, such as humic and fulvic acids. The color of potable waters is typically determined by visually comparing a water sample to known color solutions prepared from a standard platinum-cobalt solution. Old water under anaerobic conditions may appear black or gray in color due to the presence of metallic sulfides.

Odor

The generation of gases during decomposition of organic matter or reduction of dissolved sulfate often creates odorous compounds. Most odorous gases, such as hydrogen sulfide and the sulfur-bearing mercaptans, are formed under anaerobic conditions, so providing adequate dissolved oxygen is the first step toward controlling odors. The control of odors is among the priority issues with respect to public acceptance of a project.

Dissolved Oxygen

Adequate dissolved oxygen in surface waters is required for aerobic respiration and is needed to protect fish and other aquatic life. The presence of dissolved oxygen (DO) leads to oxidizing conditions that minimize the formation of noxious odors and prevents the solubilization of certain metals (e.g., iron, manganese); however, introduction of DO into anaerobic or reduced aquifers may oxidize sulfide minerals and increase the release of metals. The inverse situation may occur as well; MUS activities that low DO water into a previously oxidized part of the aquifer may lead to reductive dissolution of minerals and the release of metals.

pH

The hydrogen ion concentration is an important quality parameter for all waters. The usual means of expressing hydrogen ion concentration is pH, which is defined as the negative logarithm of the hydrogen ion concentration. pH influences the surface charge on solid surfaces, the distribution of acidic and basic compounds, the form of a chemical in solution, the solubility of compounds, the physical shape of organic molecules, and the toxicity of the medium.

Oxidation-Reduction p\Potential (ORP)

Eh is another critical parameter because of the effect of high Eh waters on iron-bearing minerals. Such solutions, which often contain high levels of dissolved oxygen, alter primary minerals to iron oxyhydroxides, thus changing the water chemistry as well as altering the aquifer properties. Eh and pH are also primary controls on the population of subsurface bacteria that biodegrade certain organic contaminants, as well as on those that cause illness. Some sulfate-reducing bacteria, for example, survive or thrive only in the absence of dissolved oxygen.

Specific Conductance

Specific conductance is a measure of how well a given water sample conducts an electrical current. It is defined as the "reciprocal of the resistance in ohms measured between opposite faces of a centimeter cube of an aqueous solution at a specified temperature" (Hem, 1985). It is a straightforward measurement that can be made with reasonable accuracy in the field. It is, therefore, often used as a proxy in lieu of the total dissolved solids (TDS) in a solution. The relationship between conductivity and TDS depends on the actual dissolved anions and cations (i.e., sodium chloride and calcium sulfate solutions of the same strength would have different specific conductances), so it is only a general indicator. However, if the dissolved salts are known to be of seawater origin, the correlation may be quite good. Specific conductance also depends on temperature.

Temperature

Water temperature can be important for several reasons. In the case of recharge water, it can affect the speed ("kinetics) of reactions in the subsurface. In general, higher temperatures increase the rate of most chemical reactions. Reactions involving dissolved gases (e.g., limestone dissolution, which involves dissolved carbon dioxide) are also affected by temperature. Bacteria involved in clogging and oxidation-reduction reactions (including those related to aesthetic concerns) are more or less active depending on temperature. In the case of discharge water, higher temperatures generally have lower dissolved oxygen, and this might impact use of the water for environmental purposes.

INORGANIC CONSTITUENTS

Inorganic chemical constituents of concern in MUS source waters are summarized in Table 4-2. Inorganic chemicals of concern can be grouped as nutrients, nonmetals, and metals. Nitrogen and phosphorous species are known as nutrients because they are essential for growth of microorganisms and plants. The nonmetals of concern are hydrogen ions and dissolved salts, such as chloride, sulfate, and boron. The metals of concern are often present at trace concentrations, and many are classified as priority pollutants. Examples of toxic metals include arsenic, cadmium, mercury, lead, and chromium. Iron and manganese are metals that influence the aesthetic quality of the water. The presence of inorganic constituents in excessive quantities will interfere with many beneficial uses of the water due to aesthetic issues or because of their toxicity.

Nutrients

Nitrogen and phosphorus are essential nutrients for growth of biomass. Their presence in water can stimulate growth of algae and microorganisms. This creates nuisance conditions during storage of the water and can accelerate biomass clogging in the subsurface. Nitrogen exists in several oxidation states with ammonia, nitrogen gas, nitrite, and nitrate being the common forms in water supplies. Chemical and biological reactions can convert one nitrogen form to another. Unionized ammonia (NH_3) is toxic to fish and other aquatic life. At the pH of most natural waters, ammonia is mostly in the cationic form (NH_4^+). As water containing NH_4^+ contacts soil, NH_4^+ is usually rapidly removed from solution by ion exchange processes. Nitrite is relatively unstable and is easily oxidized to nitrate. Nitrite is also toxic to fish, other aquatic life, and humans. Nitrate is the most oxidized form of nitrogen. Nitrate readily moves with water through the subsurface and can impact the quality of water on a large scale. Nitrate is limited to 10 mg/L as nitrogen in drinking water because of its serious toxicity to infants. The usual aqueous forms of phosphorus are orthophosphate, polyphosphate, and organic phosphate. Phosphorus does not undergo change in oxidation state. Phosphates tend to precipitate and be removed by ion exchange in the subsurface.

Salts

Water in contact with the earth will naturally accumulate dissolved salts, such as sodium, calcium, magnesium, potassium, chloride, sulfate, and bicarbonate. A gravimetric measurement technique that quantifies the residue of filtered water upon evaporation is termed total dissolved solids and provides an indicator of the total salt content. The TDS concentration is an important indicator of the usefulness of water for various applications. For example, drinking water has a recommended maximum TDS of 500 mg/L. Excessive dissolved salts influence the ability to recycle water in an MUS system as they impart a salty taste (aesthetic concern), accelerate corrosion of metals, form deposits, and can have a laxative effect in the case of sulfate.

Metals, Metalloids, and Other Constituents

Trace quantities of many metals (including metalloids), such as arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, and zinc are present in many waters. Organisms require most metals in trace quantities; therefore, the absence of trace metals in water is limiting to biomass growth. Many metals are classified as priority pollutants, so excessive amounts of these metals will interfere with beneficial uses of the water. Elevated iron and manganese in water imparts a metallic taste and causes staining of water fixtures.

The most prevalent toxicity from use of recycled water for irrigation is from boron. The source of boron is usually household detergents or discharges from industrial plants. A guideline limit of <0.7 mg/L of boron allows for unrestricted use of the water for irrigation of food crops. Boron levels above 3 mg/L are severely toxic to plants.

While many of the metals listed above can be toxic (see Table A-1), the metal of most widespread concern in MUS systems is arsenic. This is not only because the maximum contaminant level is low, but also because it is associated with commonly occurring iron oxyhydroxides and sulfides in the subsurface. These minerals often release arsenic in response to changes in oxidation-reduction state. Arsenic is treated in considerable detail in Chapter 4.

Radionuclides

Radionuclides are unstable atoms that change their atomic state through the process of radioactive decay (U.S. Department of Health, Education, and Welfare, 1970). Radioactive decay results in the release of alpha, beta, or gamma radiation. The emission of alpha and beta particles transforms an isotope into a different element, while the emission of gamma radiation reduces the energy level of the element. When alpha, beta, or gamma radiation passes through adjacent atoms, it can dislodge electrons from their orbit and create ionized species. This ionization and deposition of energy can damage materials and lead to deleterious effects in human tissues including mutagenic, teratogenic, and acute toxicity. Consequently, human exposures to radionuclides are stringently controlled.

Radionuclides in water supplies can be from natural or anthropogenic sources (Viessman and Hammer, 2005). Naturally occurring radioactive elements of importance in water often emit alpha particles. These elements, such as radium-226, can leach from geological formations and enter groundwater. Radioactivity from radium is also widespread in surface waters because of fallout from testing of nuclear weapons. Another source of radioactivity in water supplies is small releases from nuclear power plants and industrial users of radioactive materials (e.g., weapons manufacture and testing; medical applications; nuclear fuel processing, use, and disposal). Radionuclides are currently regulated in drinking water by the U.S. Environmental Protection Agency (EPA). The limit for gross alpha particle activity is 15 pCi/L, and the limit for the sum of ^{226}Rd and ^{228}Rd is 5 pCi/L. The measurement unit of pCi/L is 10^{-12} curie per liter, with a curie being the radioactivity of 1 gram of radium. The activity from beta radiation is primarily from nuclear weapons testing, and the allowable amount is up to 4 mrem/yr, which is a measure of equivalent absorbed dose. Uranium has several radioactive isotopes, and the uranium concentration must be less than 30 μg/L.

A study by the National Academy of Sciences (1977) concluded that natural background radiation can be estimated to cause 4.5 to 45 fatal cases of cancer per year per million people, and less than 1 percent of the risk is attributed to radionuclides in drinking water. Consequently, in most water supplies, it is not possible to measure any adverse health effects from radionuclides with certainty. Monitoring for radionuclides in water supplies is straightforward, and it is prudent to periodically monitor the radionuclide activity in the source and extracted waters in MUS systems to ensure the safety of the consumer.

ORGANIC CONSTITUENTS

Residual organic carbon is a concern in underground storage systems because some of these compounds are associated with a broad spectrum of potential health concerns (Asano, 1998). Three groups of residual organic chemicals require attention: (1) natural organic matter (NOM) present in most water supplies, (2) soluble microbial products (SMPs) formed during the wastewater treatment process and resulting from the decomposition of organic compounds (Barker and Stuckey, 1999), and (3) synthetic organic compounds (SOC) added by consumers and generated as disinfection by-products (DBPs) during the disinfection of water and wastewater.

Natural Organic Matter and Soluble Microbial Products

Natural organic matter and soluble microbial products are mixtures of compounds that cannot be effectively measured individually. When NOM and SMPs are measured as a group as dissolved organic carbon, the concentrations of organic carbon are typically measured in the milligram-per-liter range. Most waters contain NOM and reclaimed waters contain a mixture of NOM and SMPs. These compounds are not known to present significant health concerns. The primary concern above NOM and SMPs is their potential to form disinfection by-products and to stimulate biological growth in distribution systems, in wells, or in situ. Synthetic organic compounds and disinfection by-products are measured individually at concentrations of microgram or nanogram per liter. When a pool of organic carbon exists, the synthetic organic carbon compounds and disinfection by-products may represent less than 1 percent of the total organic carbon. However, concerns about both human and aquatic health effects are generally associated with SOCs and DBPs.

Most waters used in underground storage systems receive limited characterization of NOM and/or SMPs that comprise the bulk of the organic carbon compounds present. Typically, these compounds are quantified by dissolved organic carbon measurements and ultraviolet absorbance (UVA) (Ma and Yin, 2001). UVA provides a relative measure of the aromatic content of the dissolved organic carbon and serves as a predictor of disinfection by-product formation potential. DBP formation potential tests are also used to characterize the reactivity of NOM and SMPs with disinfectants. Advanced characterization of

NOM and SMPs has been done for research purposes. Separation techniques used for bulk organic carbon include molecular weight fractionation, size exclusion chromatography and fractionation based on hydrophobicity (Croue et al., 2000). Spectroscopic characterization of organic carbon isolates may be done using ^{13}C nuclear magnetic spectroscopy, Fourier transform infrared spectroscopy, and fluorescence spectroscopy (Fox et al., 2001). In addition, elemental analysis may be done to determine the elemental composition of organic compound mixtures. The majority of these techniques do not identify significant differences in organic compound structure, function, and reactivity when comparing NOM samples with mixtures of NOM and SMPs in reclaimed waters. SMPs do have elevated levels of organic nitrogen and associated fluorescence compared to NOMs.

Organic compounds are removed during subsurface storage by a combination of filtration, sorption, oxidation-reduction and biodegradation. Biodegradation is the primary sustainable removal mechanism for organic compounds during subsurface transport. The concentrations of NOM and SMPs are reduced during subsurface transport as high-molecular-weight compounds are hydrolyzed into lower-molecular-weight compounds and the lower-molecular-weight compounds serve as substrate for microorganisms. As the concentrations of NOM and SMPs are decreased, the disinfection by-product potential associated with these compounds is also decreased (AwwaRF, 2001). Synthetic organic compounds at concentrations too low to directly support microbial growth may be co-metabolized as NOM and SMPs serve as the primary substrate for growth. Given sufficient surface area and contact time, the water used for underground storage may approach the quality of native groundwaters with respect to organic carbon content.

Total Organic Carbon

The performance of sustainable underground storage systems with respect to organic carbon transformations has often been quantified by measuring total organic carbon (TOC). While total organic carbon does not provide any significant information regarding health effects, it has often been used as a surrogate for organic carbon removal for several reasons. TOC is simple to measure, and most laboratories can measure it rapidly and accurately. The second reason is that 1 mg of organic carbon may be composed of millions of different compounds and these compounds cannot be individually quantified. Furthermore, if all these compounds could be quantified, using the data collected would be very difficult to interpret. A health effects study completed in Los Angeles County determined there were no impacts of groundwater recharge from reclaimed water. The California Department of Health Services estimated that the maximum TOC concentration from reclaimed water in the drinking water supply was 1 mg/L. Presently, California has a guideline of 0.5 mg/L of TOC from recycled water that may be used for drinking water. The State of Washington has a proposed rule of 1 mg/L of TOC for direct injection with reclaimed water and Florida has limit of 3 mg/L of TOC for groundwater recharge with reclaimed water.

When TOC concentrations for reclaimed water are set at 1 mg/L or less, the target that concentration is below the TOC concentration of most surface water sources. The result is there must be significant dilution of the reclaimed water used for recharge, or extensive treatment of the reclaimed water is required. Since most reclaimed waters contain refractory NOM and SMPs in excess of 2 mg TOC/L, treatment by nanofiltration or reverse osmosis is necessary to reduce the TOC concentration to below 1 mg/L.

Efforts to develop surrogates for organic carbon removal other than TOC are currently ongoing and were summarized at the Water Reuse Foundation Research Conference in May 2006. The fact that TOC concentrations below 1 mg/L can still contain elevated concentrations of synthetic organic compounds with significant health concerns such as *N*-Nitrosodimethylamine (NDMA). Since refractory TOC concentrations depend on the original drinking water source and SMP production during wastewater treatment, a measure of microbial activity was suggested as a surrogate during subsurface transport where the major sustainable removal mechanism is biodegradation. The mosquito repellent DEET (*N, N*-diethyl-m-toluamide, a personal care product, was suggested as a marker since it is biodegradable and almost all other biodegradable compounds were removed before DEET.

Synthetic Organic Carbon

Many waters used in underground storage systems are analyzed for contaminants regulated by the Safe Drinking Water Act, which includes maximum contaminant levels (MCLs) for 51 synthetic organic compounds. Since many of the compounds were regulated by the Safe Drinking Water Act because of groundwater contamination issues, these compounds are not often detected in water sources being considered for underground storage. Exceptions to the case include chlorinated disinfection by-products and specific agricultural chemicals. When reclaimed waters are used as a water source, monitoring for emerging contaminants of concern may be applied in specific states such as California.

Trace Organics

Trace organics in both wastewater and surface waters impacted by human and animal waste streams are of interest if used for MUS systems. The behavior of selected trace organics during underground storage has been studied to identify and quantify processes that affect organic contaminant attenuation during subsurface transport. The majority of this research has focused on the use of reclaimed water as a source water and also the use of sewage-contaminated surface waters in bank filtrations systems. For trace organic compounds, the principal attenuating processes are biological transformation and sorption. The occurrence of these processes differs depending on compound structure, soil, and biogeochemical conditions. The capacity of biodegradation and sorption to attenuate organic contaminants may vary considerably depending upon the underground storage system. Geochemical factors such as changes in redox conditions and the formation of organic complexes may also affect transformations during subsurface transport. Attempts to develop time-distance relationship for the attenuation processes have been successful for specific types of systems such as flow through porous media in sand and gravel aquifers. Initial research activities focused on the fate of compounds present at microgram-per-liter concentrations in source waters. These compounds included clofibric acid, surfactants such as alkylphenolethoxylates, disinfection by-products, nitrilotriacetic acid, and ethylenediaminetetraacetic acid (EDTA). As analytical techniques improved to detect compounds at nanogram per liter concentrations, concerns about pharmaceuticals and personal care products (PPCPs) and endocrine disrupting compounds (EDC) led to additional research on these emerging contaminants of concern. As a result of this research, certain anthropogenic compounds have been determined to be persistent in most underground storage systems; however, the health effects associated with these compounds at nanogram-per-liter concentrations have not been assessed. Emerging disinfection by-products such as N-nitrosodimethylamine have also been studied during subsurface transport.

The occurrence and significance of anthropogenic compounds in surface waters impacted by wastewater discharges in the United States was described in a survey by Kolpin et al. (2002), who conducted a survey of 139 streams in the United States for 93 organic waste contaminants, and a wide range of pharmaceuticals and personal care products were measured creating concerns over the safety of surface waters as drinking water supply. The widespread occurrence of these compounds in the United States and Europe was discussed by Daughton and Ternes (1999), who suggested that impacts to aquatic life and other environmental impacts were possible; however, the concentrations of pharmaceuticals were too low to have a defined impact on human health. Nevertheless, concern over these emerging contaminants has resulted in active research on the fate and transport of these compounds in the environment. Research on the fate of these compounds during bank filtration has been occurring to a large extent in Europe and to a lesser extent in the United States (Heberer et al., 2001). Several monitoring studies carried out in Berlin, Germany, between 1996 and 2000 identified pharmaceuticals such as clofibric acid, diclofenac, ibuprofen, propyphenazone, primidone and carbamazepine at individual concentrations up to the nanogram-per-liter level in influent and effluent samples from WWTPs and in all surface water samples collected downstream from the waste water treatment plants (Heberer, 2002). The persistence of carbamazepine has led researchers to suggest that carbamazepine be used as a universal indicator of anthropogenic contamination.

MICROORGANISMS

The microorganisms of concern associated with public health risks in groundwaters and source waters used for MUS may be divided into two major categories; those that are associated with fecal pollution of ground- and surface water and those that may be naturally-occurring. Groundwater has been shown to be a major source associated with waterborne disease outbreaks in the United States, because it is often used without treatment. In addition as already mentioned, the nature (perceived as protected) of groundwater and its use (as a potable supply) dictates the absence of "indicator" organisms and pathogens associated with contamination and public health risk as well as absence of pathogens.

All surface waters will have some level of algae, bacteria, and parasites; if there are sewage inputs there will also be enteric viruses and other microbes of fecal origin. Thus unless this water is pretreated to drinking water standards or infiltration systems are used to effectively remove some percentage of the microorganisms, then the source or stored water will contain these microbes. The native groundwater could also contain some bacteria and protozoa that may or may not pose a risk to human health. The targeted microbial contamination level associated with acceptable risks would depend on the use of the recovered water. For potable purposes a maximum contaminant level goal (MCLG) of zero is the target for those microorganisms associated with disease. Finally, both regrowth and attenuation (usually due to die-off) of bacteria and the free-living protozoa can occur depending on the conditions. For enteric viruses and protozoa long-term survival is of concern and interest. As a part of the attenuation via filtration or dilution (diffusion) the concentrations of the microorganisms which may migrate and be transported into other aquifers has also been an area of research.

Monitoring for the wide range of microorganisms in source, stored and recovered water has not been widely implemented. Thus there is often a presumption of microbial water quality based on the monitoring of selected "indicator" species. As mentioned the primary research has focused on drinking water MUS systems and thus those microbes associated with fecal pollution and standards and rules for potable water have been the target of most of the controversy and studies. The bacterial pathogens are rarely monitored for, a select group of viruses may be monitored for on occasion, and the protozoa are monitored for in surface waters but not generally groundwaters. None of these groups of microbes are monitored for in reclaimed waters on a routine basis (the exception being Florida which requires monitoring of *Cryptosporidium* and *Giardia* in reclaimed wastewaters).

Table A-1 gives the list of some of the routine and emerging microorganisms that may be of concern in MUS systems. These all have an MCLG of zero in drinking water.

TABLE A-1 Pathogens and Microorganisms of Possible Concern with MUS Systems

Microbe	Sources	Relationship to Groundwater	Occurrence in Water	Other Key Points
Bacteria				
Cyanobacteria	Naturally-occurring In Surface Waters	Unknown could be stable in MUS aquifers	Toxins found in surface waters alone with seasonal blooms	Not routinely monitored for yet in MUS. On the CCL
Campylobacter	Animal/ bird and human fecal wastes	Associated with groundwater outbreaks	Aerotolerant Stable in non-disinfected waters	Can cause Guillian Barré Syndrome
Arcobacter	Possibly animal and human wastes	Unknown Identified in groundwater with massive contamination	Grows at low temperatures	Emerging human pathogen
Helicobacter	Human sewage	Association with human disease and exposure to groundwater	Detected by PCR in sewage and groundwater	Emerging cause of ulcers and cancer; on the CCL
Legionella	Naturally-occurring in aquatic environment	Found in groundwater and cause of the distribution system seeding	Grows along with increased amoeba and heterotrophs in biofilms	No studies on enhancement of populations under MUS
Viruses[a].				
Adenoviruses	Human Wastewater		Detected in ground-, surface, and drinking water	High concentrations in sewage, long-term survival in water; more resistant to UV
Coxsackie	Human wastewater		CB5 found to be one of the most prevalent viruses in sewage and polluted waters.	Cause of chronic diseases.

table continues

Table A-1 Continued

Microbe	Sources	Relationship to Ground-water	Occurrence in Water	Other Key Points
Noroviruses	Human wastewater	Associated with water-borne out-breaks in groundwaters in particular	Found in sew-age, surface and ground-water	World-wide in-crease of occur-rence, no culti-vatable methods available.
Polyoma viruses (Cause of Brain and colorectal cancers and urinary tract disease)	Human wastewater		Feces, urine, and sewage	Survives in water but oral transmis-sion is uncertain
Parasites				
Cryptospori-dium	Human and animal wastes	Found in groundwaters highly	Surface water ranges from 3 to 80 percent	Chlorine-resistant; monitoring re-quired for drink-ing water sys-tems using sur-face water as part of the
Giardia	Human and animal wastes	Influenced by surface wa-ters	Surface wa-ters; 100% in sewage	LTESWTR
Naegleria	Free-living protozoan in water	Recently associated with ground-water in Ari-zona	Occurrence is 20 percent in groundwater in Arizona found in areas with high HPC bacteria	Caused two deaths from groundwater in Arizona

NOTE: CCL = Contaminant Candidate List ; LTESWTR = Long-term Enhance Surface Water Treatment Rule; PCR = polymerase chain reaction
[a] All viruses are bio-nanoparticles and are able to move into aquifers and remain stable.

Microbes of Fecal Origin

Many microbial pathogens associated with waterborne disease are not native to the water bodies in which they are found and most have been introduced by human activities via point and nonpoint sources of fecal pollution (NRC, 2004). Thus, most surface waters and wastewaters (including reclaimed waters unless highly treated; e.g., membranes) will likely contain some concentration of microorganisms of fecal origin. Those such as the bacteria and protozoa are found in waters from human sources, animal agriculture, and wildlife, including birds, while the enteric viruses are associated with human fecal sources. Few studies exist on the ecology and evolution of microbial pathogens in environmental waters in comparison to research investigating their pathogenicity or health impacts. To gain a real understanding of their abundance, distribution, and fate in the environment, more monitoring studies tied to testing of specific hypotheses and application of advanced microbiological tools should be undertaken.

Waterborne diseases (of a fecal-oral nature) in the United States that plagued populations in the 1800s and early 1900s such as cholera and typhoid, are no longer a major concern, due mainly to wastewater treatment, water filtration, and disinfection processes instigated since the early 1900s. Currently, the enteric pathogens of concern originating from sewage and animal waste include bacteria such as *Arcobacter, Campylobacter, Helicobacter,* and pathogenic *Escherichia coli*, protozoa such as *Cryptosporidium* and *Giardia,* and the human enteric viruses. These are described briefly below.

Bacterial Waterborne Microbial Pathogens

The primary bacteria of emerging concern in water include *Arcobacter,* pathogenic types of *Escherichia coli*, *Campylobacter,* and *Helicobacter.* These come from animals and human fecal material, survive particularly well in groundwater, and may regrow; they have been associated with waterborne disease in groundwater systems.

Campylobacter is found in a wide range of mammalian and avian hosts, including cows, sheep, pigs, chickens, and crows. *Campylobacter* is a major cause of bacterial diarrheal illness and is often associated with groundwater outbreaks. The bacteria are found in waters influenced directly by human and/or animal wastes. In addition to having a variety of hosts, *Campylobacter* cells are viable for months. The most recent groundwater outbreaks in which some of the cases were due to *Campylobacter* include Walkerton, Ontario (Clark et al., 2003), and Put-in-Bay, Ohio, in Lake Erie (Fong et al., 2007). *Campylobacter* are believed to originate from surface waters contaminated with animal and human wastes.

Arcobacter spp. belong to the same family as *Campylobacter* and were formerly classified as *Campylobacter cryaerophila*. *Arcobacter* spp. (which include *Arcobacter butzleri*, *A. cryaerophilus*, or *A. skirrowii* strains) were reported as the fourth most common *Campylobacter*-like organisms isolated from stool specimens in Belgium between 1995 and 2002 (Vandenberg, 2004). Their morphology closely resembles *Campylobacter* and *Helicobacter*. They share the characteristics of being fastidious, Gram-negative, motile by means of flagella, and spiral shaped (Wesley, 1997). A distinctive feature that differentiates *Arcobacter* spp. from *Campylobacter* spp. and *Helicobacter* spp. is that *Arcobacter* spp. can grow at 15 °C and are aerotolerant (Wesley, 1997). *Arcobacter* spp. were first isolated from aborted bovine and porcine fetuses in 1977 (Ellis et al., 1977). Recent studies suggest that *Arcobacters*, especially *A. butzleri*, may be associated with persistent, watery diarrhea and bacteremia (Vandenberg, 2004). Little is known about the mechanisms of pathogenicity, potential virulence, or clinical importance of *Arcobacter* because these organisms are often misidentified as *Campylobacter* if specific testing to species level is not done (Diergaardt et al. 2004). In a recent waterborne outbreak in Lake Erie, 41 percent of the wells were positive for *A. butzleri*. This bacterium should be considered an emerging waterborne pathogen and monitored in MUS systems where disinfection is not routinely used because of its high prevalence in water samples and its ability to replicate at groundwater temperatures (Fong et al., 2006).

Helicobacter pylori is a ubiquitous microorganism infecting half of the world's population (Feldman et al., 1997). A gram-negative, microaerophilic bacterium, it has been recognized as the primary cause of peptic ulcers, chronic gastritis, MALT lymphoma and stomach cancer. The World Health Organization has classified it as a class I carcinogen (Aruin, 1997; Blaser, 1990). About 50 percent of the U.S. population is thought to be symptomatic or asymptomatic carriers, even though the source of human infection is not well understood. Water supplies contaminated with fecal material may be a potential source of *H. pylori* transmission (Hulten et al., 1996). Rolle-Kampczyk et al. (2004) found a significant correlation between well water contaminated with *H. pylori* detected by PCR [polymerase chain reaction] and colonization status in humans.

Protozoan Parasites

The protozoan parasites in surface water are associated with fecal pollution from both humans and animals. Their survival is greater than the bacteria but they can not regrow. They produce egg-like structures that are relatively large and thus are removed during drinking water filtration and infiltration. The risk to MUS systems is from the storage of surface waters and reclaimed waters and the recovery and use of the waters where humans may be exposed via drinking or recreation.

Giardia is the most commonly isolated intestinal parasite in the world (Gardner and Hill, 2001). *Giardia* cysts are present in high numbers in domestic sewage and are of particular concern due to their inherent resistance to disinfectants commonly used in wastewater treatment processes (Rose et al., 1996). *Cryptosporidium* is an intestinal parasite also found worldwide. The oocysts have been detected in untreated wastewater and also in some drinking water sources (Smith and Rose, 1998; Rose et al., 2002). *Cryptosporidium* oocysts are completely resistant to the levels of chlorine commonly used in wastewater and drinking water treatment. *Cryptosporidium* caused the largest waterborne outbreak ever documented in the United States, where more than 400,000 people became ill in Milwaukee, Wisconsin, when a drinking water treatment plant malfunctioned (MacKenzie et al., 1994).

These parasites are considered surface water contaminants associated with fecal material from animals and sewage from humans. Parasite occurrence in groundwater in a national survey was found 8 to 20 percent positive, based on susceptibility of the aquifer (sandy aquifers, horizontal wells) (Hancock et al., 1998), and under the Surface Water Treatment Rule, the detection of parasites in groundwater indicates legally a groundwater "under the influence" that must be treated as a surface water for potable purposes.

Human Enteric Viruses

The enteric viruses of concern with MUS systems originate from human feces; thus, contaminated surface waters, wastewaters, and septic tank effluents are the sources. They survive in water, particularly groundwater, and as bio-nanoparticles are readily transported in aquifers similar to other conservative dissolved chemicals. They do not regrow.

It is estimated that more than 100 human enteric viruses can be transmitted by human feces (Mara and Horan, 2003). These viruses infect the gastrointestinal tract and are transmitted via person-to-person contact, and exposure to contaminated food or water. The viruses known to be present in relatively large numbers in human feces include the cultivable enteroviruses (i.e., echoviruses, coxsackieviruses), adenoviruses, reoviruses, rotaviruses, Hepatitis A virus, and Norwalk-like viruses. Viruses are of particular concern when present in low numbers even in reclaimed wastewater because of their characteristically low (<10) infectious dose (Haas et al., 1999).

Typically, viruses tend to be more infective and decay more slowly than bacteria. However, the prevalence and concentration of the hundreds of different enteric viruses in water varies with the health of the community and the type of wastewater treatment system. For example, the recent massive outbreaks of *Norovirus*, a waterborne pathogen found in sewage, would be associated with increased concentrations of this virus in the wastewater collection system. Because *Norovirus* is not an enterovirus, it is not typically monitored for and can not be measured using the "standard" virus testing employed.

There have been a number of national groundwater surveys. Viruses have been detected in groundwater ranging from 2 to 30 percent depending on the study (Fout et al.). At least in one study 8% of the wells were positive by cultivation methods and 30 percent were positive for viruses by genetic methods (Abbadezegan et al., 2003).

Indicator Organisms

Coliform bacteria have a long history of use as indicators of microbiological water safety associated primarily with fecal contamination potential. Yet the deficiencies of the indicator system are well recognized (NRC, 2004) with regard to key pathogens of concern including protozoa, viruses, and some other bacteria, particularly naturally occurring bacteria such as *Legionella*. In addition to routine monitoring of coliform bacteria, several alternative monitoring approaches have been suggested to provide a tool box of indicators with various survival, source associations, and transport characteristics. These alternative indicators include Enterococci, *Clostridium perfringens*, and F-specific coliphages.

The coliform bacteria (TC) are rod-shaped, gram-negative, heterotrophic bacteria that range in length from 0.5 to 2 µm. Coliforms are facultative anaerobes, capable of aerobic respiration, anaerobic respiration, and fermentation pathways for ATP (adenosine triphosphate) synthesis. Total coliforms consist of lactose-fermenting bacteria that do not form spores and that grow at 37oC. They are used as the drinking water standard and for groundwater designated as potable, and should be negative in 100 mL. This group is able to grow in the water environment, and its occurrence in water has not been associated with health risks. Yet from the regulatory standpoint, this group of bacteria governs the "acceptability" for groundwater as a potable water supply.

The fecal coliform (FC) group is a subgroup of total coliforms that is defined by its ability to grow at 44.5°C. Although *Escherichia coli* is the dominant fecal coliform in the gastrointestinal tract of mammals, members of other genera such as *Citrobacter* and *Klebsiella* can meet the operational definition of fecal coliform (LeClerc et al., 2001). The enterococci are spherical, Gram-positive aerotolerant bacteria that do not utilize oxygen for ATP synthesis. Fecal coliform bacteria are used for wastewater discharge in many states, and *E. coli*, as a specific member of the FC group, is often used for recreational waters. The total coliform rule explicitly states that TC plus FC or *E. coli* should be absent from drinking water, with the later groups affiliated with mandated boil orders if found.

Enterococci are a subgroup of fecal streptococci and tend to be more persistent than fecal coliforms, particularly through wastewater treatment processes, and have been better associated with groundwater contamination and disease risks from septic systems (Borchardt et al, 2003). *Clostridium perfringens* are rod-shaped, obligately anaerobic, Gram-positive opportunistic pathogens that tend to survive longer in the environment than other bacteria due to the formation of endospores. It has been suggested that *C. perfringens* could be used as a surrogate indicator for protozoan pathogens because of their spore-forming capacity and resistance to disinfection.

Coliphages are viruses that infect and replicate in coliform bacteria; thus, they can only proliferate when the host bacterium is present. Coliphages can be classified in terms of whether their genome is composed of DNA or RNA. In addition they can be classified as either male specific (MS) or somatic. MS coliphages require a host that produces the fertility fimbriae (F+). Conversely, somatic coliphages bind to receptors located on the host cell wall and are not restricted to F+ hosts. Because bacteria in biological treatment systems are not in a logarithmic growth phase, it is unlikely that MS coliphages can replicate during treatment. Similarities in the physical structure, morphology, and nucleic acid composition of certain coliphages and human enteric viruses suggest that reduction of coliphages and enteric viruses may follow similar patterns, depending on the dominant physical, chemical, and/or biological removal mechanisms. In addition, the presence and survival of coliphages is related to bacterial concentrations. Coliphage testing has been proposed as a surrogate indicator for pathogens in wastewater, due in part to the failure of coliform bacteria to correlate with pathogens in wastewater (Havelaar et al., 1993). Two *E. coli* hosts were used in this study to quantify different subsets of coliphages: *E. coli* ATTC strain 700891 (F_{amp}) for male-specific RNA coliphages and *E. coli* ATCC strain 15597 for somatic and male-specific coliphages.

Despite having this array of indicators available, no one indicator or combination of indicators has adequately described the contamination risks associated with pathogenic viruses and protozoa. The viruses have always been a major concerning groundwater because of their small size, transportability in groundwaters, ability to survive in the water environment, and low infectious dose. The parasites on the other hand have been used to classify groundwaters as "under the influence" of surface waters, and if found would mandate treatment as a surface water (including filtration and disinfection).

Naturally Occurring Microorganisms Associated With a Public Health Risk

Legionella

Legionellae bacteria are ubiquitous in the freshwater aquatic environment and are prevalent in engineered water systems, including cooling towers, water heaters, and plumbed-system biofilms (Fiore *et al*., 1998; Keller *et al*., 1996; Breiman *et al*., 1990). While it was once thought that these bacteria were associated solely with surface waters, it is now clear that groundwaters are also a source of the *Legionellae* resulting in disease. A majority of the approximately 20% of all cases of Legionnaires' disease associated with recent travel are thought to be associated with drinking water systems (CDC, 2006). In addition, sporadic community acquired disease associated with this bacterium is now recognized as an important cause of pneumonia (Stout, 1997).

The bacteria can be parasitic, infecting the mammalian alveolar macrophage and alveolar epithelium (Atlas, 1999), causing the disease legionellosis, which results in pneumonia, often referred to as Legionnaires' disease, and/or a flu-like illness known as Pontiac fever (*http://www.cdc.gov/ncidod/dbmd/diseaseinfo/legionellosis_g.htm; accessed December 2007*). Cases of legionellosis in the United States reported to CDC each year between 1980 and 1998 have averaged 356 cases with no trend (Fields et al., 2002). The results of one study have placed the estimated occurrence of Legionnaires' disease in the United States at 8,000-14,500 cases per year (Marston et al., 1997) which suggests that only 2.5 to 4.5 percent of cases are reported to the Center for Disease Control and Prevention (CDC), despite legionellosis being a reportable disease (Benin et al., 2002). In some cities, community-acquired legionellosis cases have increased during the past decade (Fisman et al., 2005)

The occurrence of the bacterium is associated with heterotrophic bacteria and free-living amoeba. Thus, processes that enhance the growth of these microorganisms may also be associated with increased *Legionella*. Since no monitoring is done for this bacterium in MUS systems, there are no data that can be used to address whether a change in the aquifer (redox, organic inputs, etc.) might be associated with the stimulation of the growth of *Legionella*.

Free-Living Amoeba

Naegleria fowleri is a free-living amoeba that has a global distribution and has been found predominantly in warm waters. This parasite grows in water and is associated with heterotrophic bacteria on which it feeds undergoing replication in a trophozoite form. However the parasite also has a cyst form in which it can survive in soil, or under desiccation and shows resistance to chlorination. The parasite causes an accidental infection in humans and a disease call primary amoebic meningoencephalitis (PAM), a brain inflammation, which leads to the destruction of brain tissue and is most always fatal. The amoebae-contaminated water generally enters the nose during recreational activities, and children are at greatest risk. From a public health perspective, some have suggested that PAM is rare. During 1989--2000, CDC's waterborne disease outbreak surveillance system documented 24 fatal cases of PAM in the United States (MMWR, 2004). The majority of these cases occurred during the summer months and among children. The ecological conditions that contribute to the risk of death are not well understood.

In October, 2002, the very first groundwater-potable water outbreak ever documented occurred in Arizona when two young children acquired the disease and died. Subsequently *Naegleria fowleri* was detected in samples collected from sources in homes associated with the deaths of the two children (Marciano-Cabral, 2003) some wells and in the distribution system. Because the disease is fatal and is not readily diagnosed, engineered groundwater systems particularly in warm climates must be concerned with the growth of this protozoan in their wells. Clearly, greater information is needed on the occurrence and potential for the parasite to grow under various conditions to avoid such tragedies in the future.

Other Bacteria

Pseudomonas, *Enterobacter*, *Acinetobacter*, *Klebsiella*, and *Stenotrophomonas* are other bacteria that have been shown to be opportunistic (i.e., a cause disease under the right conditions) and are widespread in aquatic systems. In some cases these bacteria may be associated with fecal wastes, but they can also grow and establish themselves in the water environment. These pathogens have commonly caused outbreaks in recreational settings and are of particular concern because many are now resistant to antibiotics.

Algal Toxins

The cyanobacteria and their toxins are surface water phenomenon that must be addressed for drinking water in the future because they are on the EPA *Contaminant Candidate List*. The use of any surface waters for MUS will have to consider the algal toxins, their stability, and the potential risks associated primarily with drinking water applications.

Microcystins are hepatotoxins produced by *Microcystis spp*. and other species of cyanobacteria. The World Health Organization has set a guideline of 1 μg/L (1,000 ng/L) for microcystin-LR in drinking water and suggested levels of about 2 to 4 μg/L and 10 to 20 μg/L (2,000 to 4,000 ng/L and 10,000 to 20,000 ng/L) for mild to moderate risks, respectively, associated with recreational waters. Work has been undertaken to begin to identify and quantify cyanobacteria and their associated toxins in waters. In a survey in Florida, the average microcystin concentrations ranged from 62 to 182 ng/L, with one sample site, Lake Monroe, containing a maximum concentration of 2,176 ng/L. Blooms and toxin levels were seasonal (in summer); this coincided with increased rainfall and water availability in which storage with MUS systems might occur. The stability of toxins in groundwater is not known.

Table A-2 List of Contaminants and their MCLs, as per the Environmental Protection Agency

Microorganisms

Contaminant	**MCLG[1] (mg/L)[2]**	**MCL or TT[1] (mg/L)[2]**	**Potential Health Effects from Ingestion of Water**	**Sources of Contaminant in Drinking Water**
Cryptosporidium	zero	TT[3]	Gastrointestinal illness (e.g., diarrhea, vomiting, cramps)	Human and fecal animal waste
Giardia lamblia	zero	TT[3]	Gastrointestinal illness (e.g., diarrhea, vomiting, cramps)	Human and animal fecal waste
Heterotrophic plate count	n/a	TT[3]	HPC has no health effects; it is an analytic method used to measure the variety of bacteria that are common in water. The lower the concentration	HPC measures a range of bacteria that are naturally present in the environment

			of bacteria in drinking water, the better maintained the water system is.	
Legionella	zero	TT[3]	Legionnaire's Disease, a type of pneumonia	Found naturally in water; multiplies in heating systems
Total Coliforms (including fecal coliform and *E. Coli*)	zero	5.0%[4]	Not a health threat in itself; it is used to indicate whether other potentially harmful bacteria may be present[5]	Coliforms are naturally present in the environment; as well as feces; fecal coliforms and *E. coli* only come from human and animal fecal waste.
Turbidity	n/a	TT[3]	Turbidity is a measure of the cloudiness of water. It is used to indicate water quality and filtration effectiveness (e.g., whether disease-causing organisms are present). Higher turbidity levels are often associated with higher levels of disease-causing microorganisms such as viruses, parasites and some bacteria. These organisms can cause symptoms such as nausea, cramps, diarrhea, and associated headaches.	Soil runoff
Viruses (enteric)	zero	TT[3]	Gastrointestinal illness (e.g., diarrhea, vomiting, cramps)	Human and animal fecal waste

Disinfection By-Products

Contaminant	MCLG[1] (mg/L)[2]	MCL or TT[1] (mg/L)[2]	Potential Health Effects from Ingestion of Water	Sources of Contaminant in Drinking Water
Bromate	zero	0.010	Increased risk of cancer	Byproduct of

				drinking water disinfection
Chlorite	0.8	1.0	Anem+ia; infants & young children: nervous system effects	Byproduct of drinking water disinfection
Haloacetic acids (HAA5)	n/a[6]	0.060	Increased risk of cancer	Byproduct of drinking water disinfection
Total Trihalomethanes (TTHMs)	none[7] ---------- n/a[6]	0.10 ---------- 0.080	Liver, kidney or central nervous system problems; increased risk of cancer	Byproduct of drinking water disinfection

Disinfectants

Contaminant	MRDLG[1] (mg/L)[2]	MRDL[3] (mg/L)[2]	Potential Health Effects from Ingestion of Water	Sources of Contaminant in Drinking Water
Chloramines (as Cl_2)	MRDLG=4[1]	MRDL=4.0[1]	Eye/nose irritation; stomach discomfort, anemia	Water additive used to control microbes
Chlorine (as Cl_2)	MRDLG=4[1]	MRDL=4.0[1]	Eye/nose irritation; stomach discomfort	Water additive used to control microbes
Chlorine dioxide (as ClO_2)	MRDLG=0.8[1]	MRDL=0.8[1]	Anemia; infants & young children: nervous system effects	Water additive used to control microbes

Inorganic Chemicals

Contaminant	MCLG[1] (mg/L)[2]	MCL or TT[3] (mg/L)[2]	Potential Health Effects from Ingestion of Water	Sources of Contaminant in Drinking Water
Antimony	0.006	0.006	Increase in blood cholesterol; decrease in blood sugar	Discharge from petroleum refineries; fire retardants; ceramics; electronics; solder
Arsenic	0[7]	0.010 as of	Skin damage or problems with circulatory	Erosion of natural deposits; runoff from

		01/23/06	systems, and may have increased risk of getting cancer	orchards, runoff from glass & electronic-sproduction wastes
Asbestos (fiber >10 micrometers)	7 million fibers per liter	7 MFL	Increased risk of developing benign intestinal polyps	Decay of asbestos cement in water mains; erosion of natural deposits
Barium	2	2	Increase in blood pressure	Discharge of drilling wastes; discharge from metal refineries; erosion of natural deposits
Beryllium	0.004	0.004	Intestinal lesions	Discharge from metal refineries and coal-burning factories; discharge from electrical, aerospace, and defense industries
Cadmium	0.005	0.005	Kidney damage	Corrosion of galvanized pipes; erosion of natural deposits; discharge from metal refineries; runoff from waste batteries and paints
Chromium (total)	0.1	0.1	Allergic dermatitis	Discharge from steel and pulp mills; erosion of natural deposits
Copper	1.3	TT[8]; Action Level=1.3	Short term exposure: Gastrointestinal distress Long term exposure: Liver or kidney damage People with Wilson's Disease should consult their personal doctor if the amount of copper in their water exceeds the action level	Corrosion of household plumbing systems; erosion of natural deposits

Cyanide (as free cyanide)	0.2	0.2	Nerve damage or thyroid problems	Discharge from steel/metal factories; discharge from plastic and fertilizer factories
Fluoride	4.0	4.0	Bone disease (pain and tenderness of the bones); Children may get mottled teeth	Water additive which promotes strong teeth; erosion of natural deposits; discharge from fertilizer and aluminum factories
Lead	zero	TT[8]; Action Level=0.015	Infants and children: Delays in physical or mental development; children could show slight deficits in attention span and learning abilities Adults: Kidney problems; high blood pressure	Corrosion of household plumbing systems; erosion of natural deposits
Mercury (inorganic)	0.002	0.002	Kidney damage	Erosion of natural deposits; discharge from refineries and factories; runoff from landfills and croplands
Nitrate (measured as Nitrogen)	10	10	Infants below the age of six months who drink water containing nitrate in excess of the MCL could become seriously ill and, if untreated, may die. Symptoms include shortness of breath and blue-baby syndrome.	Runoff from fertilizer use; leaching from septic tanks, sewage; erosion of natural deposits
Nitrite (measured as Nitrogen)	1	1	Infants below the age of six months who drink water containing nitrite in excess of the MCL could become seriously ill and, if untreated,	Runoff from fertilizer use; leaching from septic tanks, sewage; erosion of natural deposits

			may die. Symptoms include shortness of breath and blue-baby syndrome.	
Selenium	0.05	0.05	Hair or fingernail loss; numbness in fingers or toes; circulatory problems	Discharge from petroleum refineries; erosion of natural deposits; discharge from mines
Thallium	0.0005	0.002	Hair loss; changes in blood; kidney, intestine, or liver problems	Leaching from ore-processing sites; discharge from electronics, glass, and drug factories

Organic Chemicals

Contaminant	MCLG[1] (mg/L)[2]	MCL or TT[1] (mg/L)[2]	Potential Health Effects from Ingestion of Water	Sources of Contaminant in Drinking Water
Acrylamide	zero	TT[9]	Nervous system or blood problems; increased risk of cancer	Added to water during sewage/ wastewater treatment
Alachlor	zero	0.002	Eye, liver, kidney or spleen problems; anemia; increased risk of cancer	Runoff from herbicide used on row crops
Atrazine	0.003	0.003	Cardiovascular system or reproductive problems	Runoff from herbicide used on row crops
Benzene	zero	0.005	Anemia; decrease in blood platelets; increased risk of cancer	Discharge from factories; leaching from gas storage tanks and landfills

Benzo(a)pyrene (PAHs)	zero	0.0002	Reproductive difficulties; increased risk of cancer	Leaching from linings of water storage tanks and distribution lines
Carbofuran	0.04	0.04	Problems with blood, nervous system, or reproductive system	Leaching of soil fumigant used on rice and alfalfa
Carbon tetrachloride	zero	0.005	Liver problems; increased risk of cancer	Discharge from chemical plants and other industrial activities
Chlordane	zero	0.002	Liver or nervous system problems; increased risk of cancer	Residue of banned termiticide
Chlorobenzene	0.1	0.1	Liver or kidney problems	Discharge from chemical and agricultural chemical factories
2,4-D	0.07	0.07	Kidney, liver, or adrenal gland problems	Runoff from herbicide used on row crops
Dalapon	0.2	0.2	Minor kidney changes	Runoff from herbicide used on rights of way
1,2-Dibromo-3-chloropropane (DBCP)	zero	0.0002	Reproductive difficulties; increased risk of cancer	Runoff/leaching from soil fumigant used on soybeans, cotton, pineapples, and orchards
o-Dichlorobenzene	0.6	0.6	Liver, kidney, or circulatory system	Discharge from industrial chemical

			problems	factories
p-Dichlorobenzene	0.075	0.075	Anemia; liver, kidney or spleen damage; changes in blood	Discharge from industrial chemical factories
1,2-Dichloroethane	zero	0.005	Increased risk of cancer	Discharge from industrial chemical factories
1,1-Dichloroethylene	0.007	0.007	Liver problems	Discharge from industrial chemical factories
cis-1,2-Dichloroethylene	0.07	0.07	Liver problems	Discharge from industrial chemical factories
trans-1,2-Dichloroethylene	0.1	0.1	Liver problems	Discharge from industrial chemical factories
Dichloromethane	zero	0.005	Liver problems; increased risk of cancer	Discharge from drug and chemical factories
1,2-Dichloropropane	zero	0.005	Increased risk of cancer	Discharge from industrial chemical factories
Di(2-ethylhexyl) adipate	0.4	0.4	Weight loss, liver problems, or possible reproductive difficulties.	Discharge from chemical factories
Di(2-ethylhexyl) phthalate	zero	0.006	Reproductive difficulties; liver problems; increased risk of cancer	Discharge from rubber and chemical factories

Dinoseb	0.007	0.007	Reproductive difficulties	Runoff from herbicide used on soybeans and vegetables
Dioxin (2,3,7,8-TCDD)	zero	0.00000003	Reproductive difficulties; increased risk of cancer	Emissions from waste incineration and other combustion; discharge from chemical factories
Diquat	0.02	0.02	Cataracts	Runoff from herbicide use
Endothall	0.1	0.1	Stomach and intestinal problems	Runoff from herbicide use
Endrin	0.002	0.002	Liver problems	Residue of banned insecticide
Epichlorohydrin	zero	TT[9]	Increased cancer risk, and over a long period of time, stomach problems	Discharge from industrial chemical factories; an impurity of some water treatment chemicals
Ethylbenzene	0.7	0.7	Liver or kidneys problems	Discharge from petroleum refineries
Ethylene dibromide	zero	0.00005	Problems with liver, stomach, reproductive system, or kidneys; increased risk of cancer	Discharge from petroleum refineries
Glyphosate	0.7	0.7	Kidney problems; reproductive difficulties	Runoff from herbicide use
Heptachlor	zero	0.0004	Liver damage;	Residue of

			increased risk of cancer	banned termiticide
Heptachlor epoxide	zero	0.0002	Liver damage; increased risk of cancer	Breakdown of heptachlor
Hexachlorobenzene	zero	0.001	Liver or kidney problems; reproductive difficulties; increased risk of cancer	Discharge from metal refineries and agricultural chemical factories
Hexachlorocyclo-pentadiene	0.05	0.05	Kidney or stomach prob-lems	Discharge from chemical factories
Lindane	0.0002	0.0002	Liver or kidney problems	Run-off/leaching from insecti-cide used on cattle, lumber, gardens
Methoxychlor	0.04	0.04	Reproductive difficulties	Run-off/leaching from insecti-cide used on fruits, vegetables, alfalfa, livestock
Oxamyl (Vydate)	0.2	0.2	Slight nervous system effects	Runoff/leach-ing from insecticide used on apples, potatoes, and tomatoes
Polychlorinated biphenyls (PCBs)	zero	0.0005	Skin changes; thymus gland problems; immune deficiencies; reproductive or nervous system difficulties; increased risk	Runoff from landfills; discharge of waste chemicals

			of cancer	
Pentachlorophenol	zero	0.001	Liver or kidney problems; increased cancer risk	Discharge from wood preserv-ing factories
Picloram	0.5	0.5	Liver problems	Herbicide runoff
Simazine	0.004	0.004	Problems with blood	Herbicide runoff
Styrene	0.1	0.1	Liver, kidney, or circulatory system problems	Discharge from rubber and plastic factories; leaching from landfills
Tetrachloroethylene	zero	0.005	Liver problems; increased risk of cancer	Discharge from factories and dry cleaners
Toluene	1	1	Nervous system, kidney, or liver problems	Discharge from petroleum factories
Toxaphene	zero	0.003	Kidney, liver, or thyroid problems; increased risk of cancer	Runoff/leach-ing from insec-ticide used on cotton and cattle
2,4,5-TP (Silvex)	0.05	0.05	Liver problems	Residue of banned herbicide
1,2,4-Trichlorobenzene	0.07	0.07	Changes in adrenal glands	Discharge from textile finishing factories
1,1,1-Trichloroethane	0.20	0.2	Liver, nervous system, or circulatory problems	Discharge from metal degreasing sites and other factories

1,1,2-Trichloroethane	0.003	0.005	Liver, kidney, or immune system problems	Discharge from industrial chemical factories
Trichloroethylene	zero	0.005	Liver problems; increased risk of cancer	Discharge from metal degreasing sites and other factories
Vinyl chloride	zero	0.002	Increased risk of cancer	Leaching from PVC pipes; discharge from plastic factories
Xylenes (total)	10	10	Nervous system damage	Discharge from petroleum factories; discharge from chemical factories

Radionuclides

Contaminant	MCLG[1] (mg/L)[2]	MCL or TT[1] (mg/L)[2]	Potential Health Effects from Ingestion of Water	Sources of Contaminant in Drinking Water
Alpha particles	None[7] ---------- zero	15 picocuries per Liter (pCi/L)	Increased risk of cancer	Erosion of natural deposits of certain minerals that are radioactive and may emit a form of radiation known as alpha radiation
Beta particles and photon emitters	None[7] ---------- zero	4 millirems per year	Increased risk of cancer	Decay of natural and man-made deposits of certain minerals that are radioactive and may emit forms of radiation known as photons and beta radiation
Radium 226 and Radium 228 (combined)	None[7] ---------- zero	5 pCi/L	Increased risk of cancer	Erosion of natural deposits

Uranium	Zero	30 ug/L as of 12/08/03	Increased risk of cancer, kidney toxicity	Erosion of natural deposits

Notes:
[1] Definitions:
Maximum Contaminant Level (MCL) - The highest level of a contaminant that is allowed in drinking water. MCLs are set as close to MCLGs as feasible using the best available treatment technology and taking cost into consideration. MCLs are enforceable standards.
Maximum Contaminant Level Goal (MCLG) - The level of a contaminant in drinking water below which there is no known or expected risk to health. MCLGs allow for a margin of safety and are non-enforceable public health goals.
Maximum Residual Disinfectant Level (MRDL) - The highest level of a disinfectant allowed in drinking water. There is convincing evidence that addition of a disinfectant is necessary for control of microbial contaminants.
Maximum Residual Disinfectant Level Goal (MRDLG) - The level of a drinking water disinfectant below which there is no known or expected risk to health. MRDLGs do not reflect the benefits of the use of disinfectants to control microbial contaminants.
Treatment Technique - A required process intended to reduce the level of a contaminant in drinking water.
[2] Units are in milligrams per liter (mg/L) unless otherwise noted. Milligrams per liter are equivalent to parts per million.
[3] EPA's surface water treatment rules require systems using surface water or groundwater under the direct influence of surface water to (1) disinfect their water, and (2) filter their water or meet criteria for avoiding filtration so that the following contaminants are controlled at the following levels:

- Cryptosporidium: (as of1/1/02 for systems serving >10,000 and 1/14/05 for systems serving <10,000) 99% removal.
- Giardia lamblia: 99.9% removal/inactivation
- Viruses: 99.99% removal/inactivation
- Legionella: No limit, but EPA believes that if *Giardia* and viruses are removed/inactivated, *Legionella* will also be controlled.
- Turbidity: At no time can turbidity (cloudiness of water) go above 5 nephelolometric turbidity units (NTU); systems that filter must ensure that the turbidity go no higher than 1 NTU (0.5 NTU for conventional or direct filtration) in at least 95% of the daily samples in any month. As of January 1, 2002, turbidity may never exceed 1 NTU, and must not exceed 0.3 NTU in 95% of daily samples in any month.
- HPC: No more than 500 bacterial colonies per milliliter.
- Long Term 1 Enhanced Surface Water Treatment (Effective Date: January 14, 2005); Surface water systems or (GWUDI) systems serving fewer than 10,000 people must comply with the applicable Long Term 1 Enhanced Surface Water Treatment Rule provisions (e.g. turbidity standards, individual filter monitoring, Cryptosporidium removal requirements, updated watershed control requirements for unfiltered systems).
- Filter Backwash Recycling; The Filter Backwash Recycling Rule requires systems that recycle to return specific recycle flows through all processes of the system's existing conventional or direct filtration system or at an alternate location approved by the state.

[4] more than 5.0% samples total coliform-positive in a month. (For water systems that collect fewer than 40 routine samples per month, no more than one sample can be total coliform-positive per month.) Every sample that has total coliform must be analyzed for either fecal coliforms or E. coli if two consecutive TC-positive samples, and one is also positive for E.coli fecal coliforms, system has an acute MCL violation.
[5] Fecal coliform and E. coli are bacteria whose presence indicates that the water may be contaminated with human or animal wastes. Disease-causing microbes (pathogens) in these wastes can cause diarrhea, cramps, nausea, headaches, or other symptoms. These pathogens may pose a special health risk for infants, young children, and people with severely compromised immune systems.
[6] Although there is no collective MCLG for this contaminant group, there are individual MCLGs for some of the individual contaminants:

- Trihalomethanes: bromodichloromethane (zero); bromoform (zero); dibromochloromethane (0.06 mg/L). Chloroform is regulated with this group but has no MCLG.
- Haloacetic acids: dichloroacetic acid (zero); trichloroacetic acid (0.3 mg/L). Monochloroacetic acid, bromoacetic acid, and dibromoacetic acid are regulated with this group but have no MCLGs.

[7] MCLGs were not established before the 1986 Amendments to the Safe Drinking Water Act. Therefore, there is no MCLG for this contaminant.
[8] Lead and copper are regulated by a Treatment Technique that requires systems to control the corrosiveness of their water. If more than 10% of tap water samples exceed the action level, water systems must take additional steps. For copper, the action level is 1.3 mg/L, and for lead is 0.015 mg/L.
[9] Each water system must certify, in writing, to the state (using third-party or manufacturer's certification) that when acrylamide and epichlorohydrin are used in drinking water systems, the combination (or product) of dose and monomer level does not exceed the levels specified, as follows:

- Acrylamide = 0.05% dosed at 1 mg/L (or equivalent)
- Epichlorohydrin = 0.01% dosed at 20 mg/L (or equivalent)

SOURCE: Reproduced from EPA Drinking Water Contaminants. Available on the web at http://www.epa.gov/safewater/contaminants/index.html. Last accessed July 5, 2007.

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Appendix B
Committee Biographical Information

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MICHAEL WEHNER is director of water quality and technology for the Orange County Water District. He manages the R&D, water quality, laboratory, and regulatory affairs departments and is responsible for monitoring of the Orange County groundwater basin, which is the principal source of water supply for more than 2.3 million people. Wehner managed the eight year, $10 million Santa Ana River Water Quality and Health Study to investigate the water quality and health implications of using effluent dominated Santa Ana River water for groundwater recharge. The district has operated a seawater intrusion barrier with recycled water since 1976 and is currently constructing the Groundwater Replenishment System that will provide 70 million gallons per day of reverse osmosis treated wastewater for injection into the barrier and for spreading in percolation basins. Wehner has over 30 years experience in water quality control and has served on numerous panels and committees for the California Department of Health Services, the National Water Research Institute, the WateReuse Foundation, and the American Water Works Research Foundation.