EPA/600/R-21/037F | June, 2021 | www.epa.gov/research

United States
Environmental Protection
Agency

Enhanced Aquifer Recharge of Stormwater in the United States:

State of the Science Review

Office of Research and Development
Washington, DC

EPA/600/R-21/037F
June 2021
www.epa.gov/research

Enhanced Aquifer Recharge of Stormwater in the United States: State of the Science Review

Center for Public Health and Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC 20460

DISCLAIMER

QUALITY ASSURANCE SUMMARY

This work was conducted under the Agency's Quality Assurance (QA) program for environmental information, with an approved Quality Assurance Project Plan for Issue Paper on Enhanced Aquifer Recharge Using Stormwater, L-HEEAD-0032809-QP-1-1. Independent QA audits were not deemed necessary, product was reviewed by QA, two internal technical reviewers and by external Peer Reviewers.

Preferred citation:
U.S. EPA (Environmental Protection Agency). 2021. Enhanced Aquifer Recharge of Stormwater in the United States: State of the Science Review. Office of Research and Development, Washington, DC; EPA/600/R-21/037F. Available online at http://www.epa.gov/research.

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ACRONYMS AND ABBREVIATIONS

ASR	aquifer storage and recovery
BMP	best management practice
BTEX	benzene, toluene, ethylbenzene, and xylene
cfs	cubic feet per second
EAR	enhanced aquifer recharge
E_h	reduction potential
EPA	Environmental Protection Agency
FIB	fecal indicator bacteria
GIS	geographic information system
GPR	ground-penetrating radar
GWPC	Ground Water Protection Council
MAR	managed aquifer recharge
MCDA	multi-criteria decision analysis
MCL	Maximum Contaminant Level
MFI	membrane filtration index
MGD	million gallons per day
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
µg/L	micrograms per liter
MS4	municipal separate storm sewer system
MTBE	methyl tert-butyl ether
ng/L	nanograms per liter
n/L	number per liter
n/100 mL	number per hundred milliliters
NASEM	National Academies of Sciences, Engineering, and Medicine
NJDEP	New Jersey Department of Environmental Protection
NMR	nuclear magnetic resonance
NPDES	National Pollutant Discharge Elimination System
NRC	National Research Council
NRMMC-EPHC	Natural Resource Management Ministerial Council, Environment Protection and Heritage Council
NSQD	National Stormwater Quality Database
NURP	Nationwide Urban Runoff Program

PAH	polyaromatic hydrocarbon
PFAAs	perfluoroalkyl acids
PFAS	per- and polyfluoroalkyl substances
PFOA	perfluorooctanoic acid
PFOS	perfluorooctane sulfonic acid
PVC	polyvinyl chloride
RPR	recharge to precipitation ratio
SSF	slow-sand filter
SWPA	source water protection area
TSS	total suspended solids
USGS	United States Geological Survey
UST	underground storage tank
UV	ultraviolet

AUTHORS

EPA's Office of Research and Development was responsible for producing this report. The report was prepared by Eastern Research Group (ERG) under EPA Contract No. EP-C-16-015. Dr. Thomas Johnson served as the Technical Directive lead, providing overall direction and technical assistance, and was a contributing author.

Eastern Research Group

Julie Blue, Ph.D., ERG
Sam Arden, Ph.D., P.E., ERG
Jason Rose, P.E., PG Environmental
Andrew Shapero, P.E., ERG

EPA ORD

Thomas Johnson, Ph.D.
Jim Carleton, Ph.D.
Michael Pennino, Ph.D.
Laurie Alexander, Ph.D.
Britta Bierwagen, Ph.D.

REVIEWERS

This report was much improved by many excellent and thoughtful comments provided by reviewers Thomas B. Boving, Ph.D.; Gretchen R. Miller, Ph.D., P.E.; and Mary L. Musick.

We are also grateful for comments on an earlier draft of this report by EPA staff members: Chris Impellitteri (Office of Water), Dave Burden (Office of Research and Development), and Justin Mattingly (Office of Water).

ACKNOWLEDGEMENTS

We acknowledge and thank Chi Ho Sham, Ph.D.; Jonathan Koplos, Ph.D.; and Mary Ellen Tuccillo, Ph.D. along with the entire team at ERG for their contributions to this report. We are also very grateful to Amanda Haddock and Ryan Jones at U.S. EPA ORD for their help with literature searches, and for posting of literature cited on EPA's Health and Ecological Research Online (HERO) web site. Finally, we thank the EPA ORD CPHEA management, communications, and web teams for their many contributions and support to produce this report.

EXECUTIVE SUMMARY

Groundwater aquifers in the United States support wide-ranging natural ecosystems and are a critical source of water; however, they are being overused or depleted in many areas. Enhanced aquifer recharge (EAR) can be a cost-effective way to replenish groundwater, ensuring both that water supplies are sustainable and that streamflow can be restored in the face of increasing population, urban development, and climate change (Bloetscher, 2015; Dillon et al., 2019; NRC, 2008). EAR has been implemented successfully in many locations globally and in support of diverse objectives. Interest in EAR has increased in the past decade, particularly in the western and southern United States as water stress has increased.

EAR can refer to practices with varying goals, site requirements, and infrastructure. Such practices include aquifer storage and recovery (ASR) of various types of recharge waters, injection of treated wastewater for other EAR applications (i.e., not ASR), recharge from surface water diversions, and other practices. As used here, EAR is used interchangeably with managed aquifer recharge (MAR), artificial recharge, anthropogenic aquifer recharge, and related terms.

In developed areas, water managers are increasingly looking to make use of unconventional water sources. The intentional recharge of aquifers using stormwater from urban residential, industrial, and commercial locations (referred to hereafter as “urban stormwater” or “stormwater”) is increasingly considered. EAR practices and infrastructure dovetail with traditional stormwater management. More generally, using stormwater for EAR also leverages stormwater as a resource, and not just a nuisance or a drainage problem. This is especially important in areas facing water scarcity. Opportunities for EAR using stormwater are also likely to increase as urbanization increases.

Using stormwater for EAR, however, also poses risks. Stormwater can contain chemical and microbial contaminants that could be detrimental to receiving aquifers (Masoner et al., 2019). While soil/aquifer systems present opportunities for natural filtering and inactivation or removal of contaminants from stormwater, there is a need for improved understanding of best practices for effective and safe EAR using stormwater in diverse development and hydrogeologic settings.

Several comprehensive reviews of EAR/MAR are available (Bouwer et al., 2008; Dillon et al., 2019; Kazner et al., 2012; Maliva, 2020; Page et al., 2016a). Less is known about EAR using stormwater than about EAR using other sources of recharge water (e.g., surface water or treated wastewater). While the methods associated with EAR are not new, our awareness of the potential environmental risks associated with them has increased. There is value in addressing the use of stormwater in these contexts.

This report is a review and synthesis of scientific and technical literature on EAR using stormwater. Our goal is an improved understanding of the scientific foundation, including knowledge gaps, leading to best practices for EAR using stormwater. More specifically, understanding of fit-for-purpose and locally appropriate uses and risks of stormwater EAR in diverse development and hydrogeologic settings will be invaluable. The report addresses the following topics:

- Common practices and infrastructure for EAR using stormwater

- Factors affecting recharge volumes achievable
- Risks of EAR using stormwater; particularly water quality degradation
- What current scientific understanding suggests about best practices for effective and safe stormwater EAR
- Key knowledge gaps that, if filled, would help advance effective and safe stormwater EAR

The report is technical and does not address policy or regulatory issues. Its focuses on EAR using stormwater. It does not discuss EAR using wastewater, treated drinking water, or other water sources. While closely related, the effects of green infrastructure on groundwater quality are only discussed to a limited extent in this review.

1 INTRODUCTION

Groundwater aquifers in the United States support-wide ranging natural ecosystems and are a critical source of water (e.g., for drinking water and irrigation), but are being overused or depleted in many areas. Enhanced aquifer recharge (EAR) can be a cost-effective way to augment water supplies, replenish aquifers and restore streamflow, ensuring that water supplies are sustainable in the face of increasing population, urban development, and climate change (Bloetscher, 2015; Dillon et al., 2019; NRC, 2008).

EAR has been implemented successfully in many locations globally and in support of diverse objectives. EAR is a broad concept: the term can refer to practices with varying goals, site requirements, and infrastructure. Such practices include aquifer storage and recovery (ASR) of various types of recharge waters, injection of treated wastewater for other EAR applications (i.e., not ASR), recharge from surface water diversions, and other practices. As used here, EAR is used interchangeably with managed aquifer recharge (MAR), artificial recharge, anthropogenic aquifer recharge and related terms.

This report focuses on one aspect of EAR, the intentional recharge of aquifers using stormwater from urban residential, industrial, and commercial locations (referred to hereafter as “urban stormwater” or “stormwater”). In developed areas, EAR dovetails with traditional stormwater management. More generally, using stormwater for EAR also better treats stormwater as a resource, and not simply a nuisance or a drainage problem. This is especially important in areas facing water scarcity. Opportunities for EAR using stormwater are also likely to increase as urbanization increases.

Using stormwater for EAR, however, also poses risks. Stormwater can contain chemical and microbial contaminants that could be detrimental to receiving aquifers (Masoner et al., 2019). Source water protection, or the protection of drinking water supplies through land management and other actions, can help manage this risk. Source water protection is also associated with additional benefits such as preserving water quality for ecological and recreational use. Nevertheless, soil/aquifer systems also present opportunities for natural filtering and inactivation or removal of contaminants from recharging stormwater.

Several comprehensive reviews of EAR/MAR are available (Bouwer et al., 2008; Dillon et al., 2019; Kazner et al., 2012; Maliva, 2020; Page et al., 2016a). ASR methods and issues, along with other MAR practices that use wells, are discussed by Maliva and Missimer (2010), and ASR systems are reviewed by Pyne (2005) as well. Less is known about EAR using stormwater than about EAR using other sources of recharge water (e.g., surface water or treated wastewater). While the methods associated with EAR are not new, our awareness of the potential environmental risks associated with them has increased. There is value in addressing the use of stormwater in these contexts.

This report is a review and synthesis of scientific and technical literature on EAR using stormwater. Our goal is an improved understanding of the scientific foundation, including knowledge gaps, leading to best practices for EAR using stormwater. More specifically, understanding of fit-for-purpose and locally appropriate uses and risks of stormwater EAR in diverse development and hydrogeologic settings will be invaluable.

The report addresses the following topics:

- Common practices and infrastructure for EAR using stormwater
- Factors affecting recharge volumes achievable
- Risks of EAR using stormwater; particularly water quality degradation
- What current scientific understanding suggests about best practices for effective and safe stormwater EAR
- Key knowledge gaps that, if filled, would help advance effective and safe stormwater EAR

The report is technical and does not address policy or regulatory issues. It focuses on EAR using stormwater. It does not discuss EAR using wastewater, treated drinking water, or other water sources. Other reviews cover these topics, including EAR using treated wastewater (Bouwer et al., 2008; Dillon et al., 2019; Kazner et al., 2012; Maliva, 2020) and EAR using surface water diversions (Bouwer et al., 2008; Maliva, 2020; NRC, 2008). While closely related, this review also has only limited discussion of the effects of green infrastructure on groundwater quality. The effects of smaller scale, green infrastructure practices such as raingardens and swales on groundwater quality are discussed in detail in U.S. EPA (2018).

The intended audiences for this report include local and state planners as well as managers and engineers engaged in the development and implementation of strategies for EAR of stormwater. Other targeted audiences include anyone charged with developing and implementing stormwater, water reuse, urban resilience, or sustainability-related policies and practices that include consideration of EAR of stormwater.

2 APPROACH

Between July and September 2020, we conducted a keyword-based literature search in three literature databases: Web of Science, Proquest, and Science Direct. Keywords were identified based on a subset of seed literature. They included single terms to capture relevant literature and boolean combinations of terms to narrow the search and screen out unrelated literature. Keywords and boolean combinations used in the keyword search are shown in Table 2-1.

Table 2-1. Keywords Used to Conduct Literature Search

KEYWORD SEARCH	
TERM	**BOOLEAN MODIFIER = "AND"**
aquifer	stormwater OR storm water
aquifer recharge	contamin*
aquifer recharge	stormwater OR storm water
aquifer storage	contamin*
aquifer storage	stormwater OR storm water
aquifer storage and recovery	
artificial recharge	
artificial recharge	contamin*
artificial recharge	stormwater OR storm water
drainage well*	stormwater OR storm water
enhanced aquifer recharge	
green AND BMP*	contamin*
green AND BMP*	recharge
groundwater replenishment	
infiltration galler*	
in-situ infiltration	stormwater OR storm water
managed aquifer recharge	
managed underground storage	
recharge basin	aquifer
recharge basin	stormwater OR storm water
recharge basin	
recharge wel*	
recharge well	contamin*
recharge well	stormwater OR storm water
recoverable water	
stormwater OR storm water	contamin*
stormwater OR storm water	microb*
stormwater OR storm water	quality
stormwater recharge OR storm water recharge	
underground injection	contamin*
underground injection	stormwater OR storm water
underground injection control	
underground storage	contamin*
underground storage	stormwater OR storm water
water banking	
water capture	
water reuse	aquifer recharge
water reuse	aquifer recharge OR aquifer storage

Results of the keyword search returned a total of more than 5,000 initial items. The titles and abstracts of keyword search results were then screened to identify a reduced set of items considered relevant to this review. Acceptance of an item was based on the following criteria:

- The paper directly addresses water quantity or quality issues related to EAR using stormwater. Papers not directly dealing with EAR/stormwater but with a research result or application of technology relevant to best practices for EAR using stormwater were included.
- Case study or site assessment/models for an EAR project in the United States were included. Case studies or site assessment studies for EAR outside the United States were only included if they had some transferable knowledge about lessons learned or best practices for EAR that could be relevant to the United States. Specifically, select studies from Australia were included because of the seminal work Australian researchers have conducted on EAR, and because of similarities in the types of contaminants expected in urban stormwater in Australia and the United States.
- Papers addressing direct wastewater injection were excluded unless their content was also potentially relevant to EAR using stormwater.

In addition to the screening criteria above, screening also considered the source (e.g., journal vs. industry publication) and publication date of the paper. More recent papers (less than five years old) were given higher priority than papers older than 20 years. Particular emphasis was placed on review papers that synthesize technical and scientific information related to aquifer recharge using stormwater. Screening resulted in initial selection of 685 items of potential relevance to this review. We later prioritized the screened literature and reviewed the highest-priority literature in depth to determine the content for each section of the report. When gaps in information were discovered, additional, targeted searches were conducted to determine if gaps could be filled. Heat maps summarizing the results of literature search and screening are shown in Figure 2-1. Note that search and screening results are directly conditional on the methods in this study and are not comprehensive. Results presented in Figure 2-1 should be considered a sample, or survey, of what is in the published literature.

(a)		Publication Date				
		2016 - 2020	2011 - 2015	2006 - 2010	2001 - 2005	2000 or earlier
Study Endpoint	Volume/Hydrology	62	36	24	8	14
	Quality - Microbial	14	12	8	0	1
	Quality - Chemical	64	61	32	19	11
	Quality - General	31	28	14	9	13
	General	72	40	20	15	35
	Aquifer - Microbial/Biofilm	2	7	2	0	0
	Economic/Policy/Decision	12	11	2	3	4

(b)		Study Focus				
		Design/Planning/Siting	Maintenance/Pre-treatment	Performance/Risk	General/Case Study	Review Paper
Study Endpoint	Volume/Hydrology	35	36	44	25	4
	Quality - Microbial	2	2	25	1	5
	Quality - Chemical	8	27	131	7	14
	Quality - General	8	18	27	8	34
	General	42	4	9	56	71
	Aquifer - Microbial/Biofilm	1	0	10	0	0
	Economic/Policy/Decision	2	0	0	11	19

(c)		Study Endpoint						
		Volume/Hydrology	Quality - Microbial	Quality - Chemical	Quality - General	General	Aquifer - Microbial	Economic/Policy
Region	General (lab/review)	45	10	65	36	58	3	8
	Northeast	11	0	2	3	4	1	0
	Southeast	11	3	17	6	16	1	2
	Midwest	2	0	2	2	0	0	0
	Great Plains North	1	0	2	0	1	0	0
	Great Plains South	5	1	3	5	11	0	1
	Northwest	5	0	1	0	4	0	1
	Southwest	33	4	24	7	39	1	5
	Australia	12	13	25	18	23	0	9
	Other International	19	4	46	18	26	5	6

(d)		Study Topic			
		Stormwater Quality	MAR Stormwater	MAR Wastewater	MAR General
Region	General (lab/review)	24	31	22	148
	Northeast	1	13	1	6
	Southeast	1	6	3	46
	Midwest	1	3	1	1
	Great Plains North	1	0	0	3
	Great Plains South	0	3	1	22
	Northwest	0	0	0	11
	Southwest	1	21	7	84
	Australia	4	50	7	39
	Other International	2	35	16	71

Figure 2-1. Heat maps showing results of literature search and screening.

The values shown are the number of studies identified as relevant to this review, grouped according to various descriptors. These values do not necessarily include all published literature (e.g., literature on certain topics, such as EAR using wastewater, is out of scope and was only included if its content was potentially relevant to stormwater capture and EAR using stormwater). Panels show (a) study endpoint versus publication date, (b) study endpoint versus focus, (c) region versus study endpoint, and (d) region versus topic.

3 EAR OF STORMWATER—METHODS/PROCESSES

"EAR of stormwater" refers to an engineered system by which stormwater is introduced into the subsurface to recharge the aquifer. This includes injection of stormwater into aquifers via wells, as well as intentional use of spreading basins, infiltration beds, and channels to recharge aquifers with stormwater. Dry wells, or vadose zone wells, are also used for stormwater EAR. Stormwater is conveyed to the vadose zone through these wells and subsequently percolates to the underlying aquifer. Aquifer recharge using stormwater has been practiced for at least seven and a half decades (Dillon, 2005; Edwards et al., 2016; UNESCO, 2005; Sasidharan et al., 2018). This section discusses common practices and infrastructure used for EAR of stormwater.

The suitability of methods for stormwater EAR depends on a variety of location-specific technical, legal, and policy considerations (Reddy, 2008):

- Land use
- Potential for clogging
- Water use restrictions
- Additional legal and regulatory considerations

Specific stormwater EAR methods include the following:

- Infiltration ponds/basins
- Infiltration ditches
- Percolation ponds
- Infiltration trenches and galleries
- Dry wells
- Infiltration pits
- Dry riverbeds
- Injection wells (including stormwater drainage wells and ASR wells)
- Permeable pavement

Some of these practices, including infiltration galleries, dry wells, and other injection wells, are generally regulated by EPA's underground injection control program.

3.1 INFILTRATION BASINS AND PONDS AND GREEN INFRASTRUCTURE

Figure 3-1 shows techniques that can be used for stormwater EAR in settings where the target aquifer is unconfined. Infiltration basins are large, often vegetated basins designed to hold runoff and allow it to infiltrate gradually into permeable soils. They are dry when there is no rainfall. Infiltration basins provide aquifer recharge, reduce downstream erosion by managing runoff volume and peak flow, and improve water quality. They also remove some pollutants when contaminants associated with particulates settle out of standing water onto the basin floor. This can eventually lead to clogging at the surface (although in general, clogging can also be caused by poor maintenance of EAR systems).

As runoff infiltrates into and passes into the subsurface, additional pollutant removal or mitigation occurs via processes such as sorption to minerals and organic matter, biogeochemical transformations (e.g., denitrification), and physical filtration of particles (and any associated contaminants) by the soil matrix. Infiltration basins provide more storage than smaller structures such as wells or trenches, but they need more land due to their large footprint. Although they clog more slowly than smaller structures such as wells, basins can fail due to poor maintenance.

Infiltration basins are often used as a best management practice to reduce pollutant loads. Massoudieh and Ginn (2008) assessed the potential for groundwater contamination from stormwater entering the subsurface through infiltration basins in California. This study built upon previous research on fate of contaminants in infiltration basins in field experiments (Barraud et al., 1999, 2005; Datry et al., 2004; Dechesne et al., 2004, 2005) and numerical models (Diaz et al., 2006; Massoudieh et al., 2004) and focused on the effects of colloidal particles and the potential enhancement of zinc, copper, and lead transport. Although facilitated metals transport by colloids is more typically a problem in contaminated surface waters, the researchers found that colloidal particles facilitated transport of lead and recommended more rigorous research in which metal concentrations in individual phases are measured at multiple subsurface locations.

Like infiltration basins, infiltration ponds (also called percolation ponds) are designed to manage runoff volume and peak flow, improve water quality, and provide aquifer recharge. However, infiltration ponds retain open water between storms. Both infiltration basins and infiltration ponds are common best management practices (BMPs) for stormwater management but may also be implemented specifically for EAR. Infiltration ditches and permeable pavement are additional widely used surface infiltration practices. While permeable pavement can be used at small scales, it is also used by municipalities or large commercial entities for large parking lots and low-traffic roads.

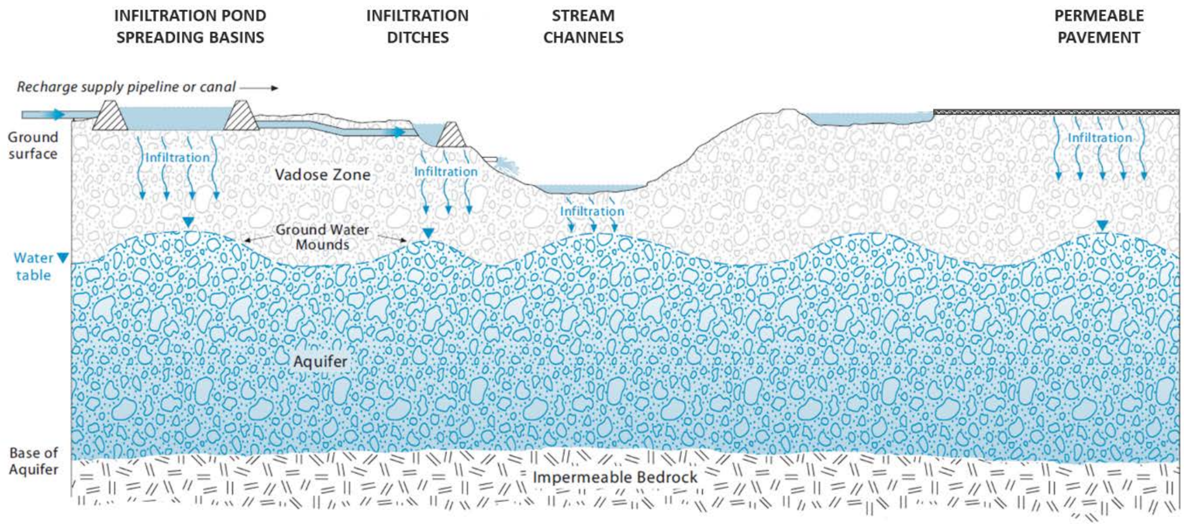

Figure 3-1. Examples of surface infiltration methodologies (adapted from Topper et al., 2004). *Surface infiltration technologies can be used for recharge into unconfined aquifers. For example, when the vadose zone and the aquifer have high vertical permeabilities, and there are no impeding layers, engineered or natural depressions can be used to recharge water from the surface to the water table. The water table then rises (Topper et al., 2004).*

3.2 DRY WELLS

Dry wells, as shown in Figure 3-2, are commonly used to manage stormwater and reduce flood risks. They are particularly well suited for EAR because they allow stormwater to bypass the low-permeability zones that may exist at some sites, resulting in increased rates of infiltration (Figure 3-2). Because dry wells infiltrate to the vadose zone (see Figure 3-2), compared to deeper injection wells that reach the saturated zone, the dry wells provide a level of attenuation of contaminants before the stormwater reaches an aquifer.

The number of dry wells needed varies with site conditions. They may not be best for sites where the vadose zone is not particularly thick, and they can be prone to clogging. Dry wells often need replacement after about five years, but they are still considered a relatively inexpensive form of stormwater EAR (Maliva, 2020; NRC, 2008) that can be more optimal than infiltration basins under certain conditions (Sasidharan et al., 2021a). Figure 3-2 shows EAR using a dry well, in addition to infiltration galleries, infiltration trenches, and infiltration pits.

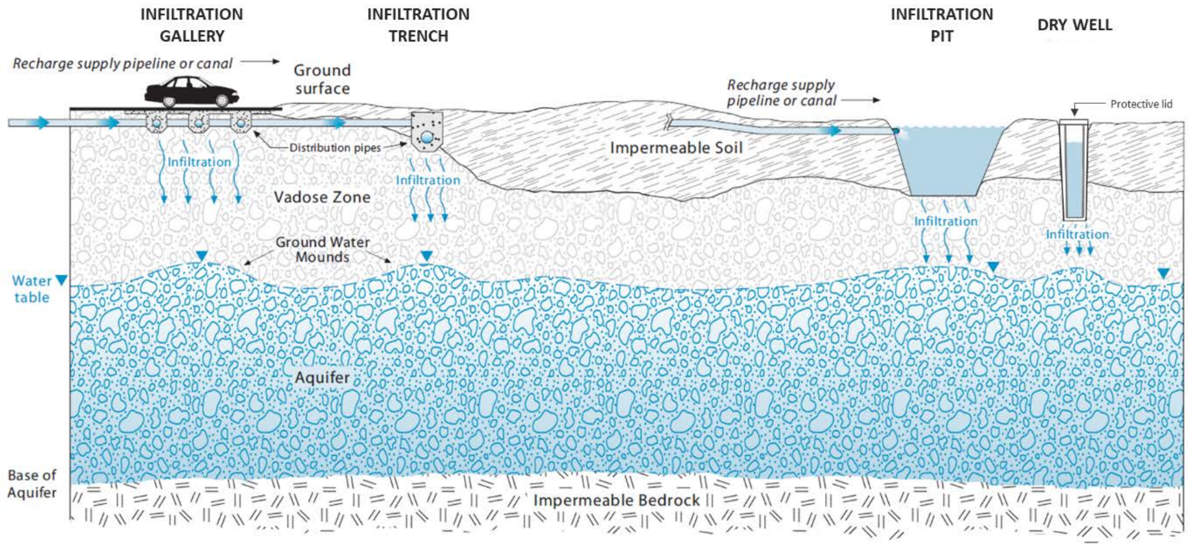

Figure 3-2. Examples of subsurface infiltration technologies (adapted from Topper et al., 2004). *At some sites, surface infiltration is not possible (e.g., because too much of the land surface is paved) and technologies that emplace water below the land surface but above unconfined aquifers are more useful (Topper et al., 2004).*

3.3 INFILTRATION TRENCHES AND GALLERIES

An infiltration trench is a linear ditch designed to collect stormwater from the surrounding area via a perforated or screened pipe and allow it to infiltrate into a highly permeable soil (Figure 3-2). These trenches are deeper than they are wide and use various designs; they may be unsupported open cuts or filled with gravel to support the sides and allow for storage. Geotextile liners may be used to separate the soil from the gravel. As well as recharging groundwater and storing runoff while it infiltrates, infiltration trenches improve water quality by removing contaminants as the runoff moves through the soil. Infiltration trenches are most useful for smaller storms due to their limited storage capacity. Infiltration trenches have the advantages of a relatively small footprint and bypassing of impermeable soil at the surface. Because infiltration trenches can fail if they become clogged with sediment, erosion control and pretreatment are important.

Infiltration galleries are wider than infiltration trenches, although there is no specific ratio of dimensions that defines a gallery. A gallery often has a series of parallel pipes to distribute the water, rather than a trench's single pipe. The fill is generally gravel, although alternate designs with plastic crates or other structures have been used.

3.4 INJECTION WELLS

Figure 3-3 and Figure 3-4 depict another common method of EAR, direct injection into an aquifer via an injection well. Injection wells are generally regulated by EPA's underground injection control program.

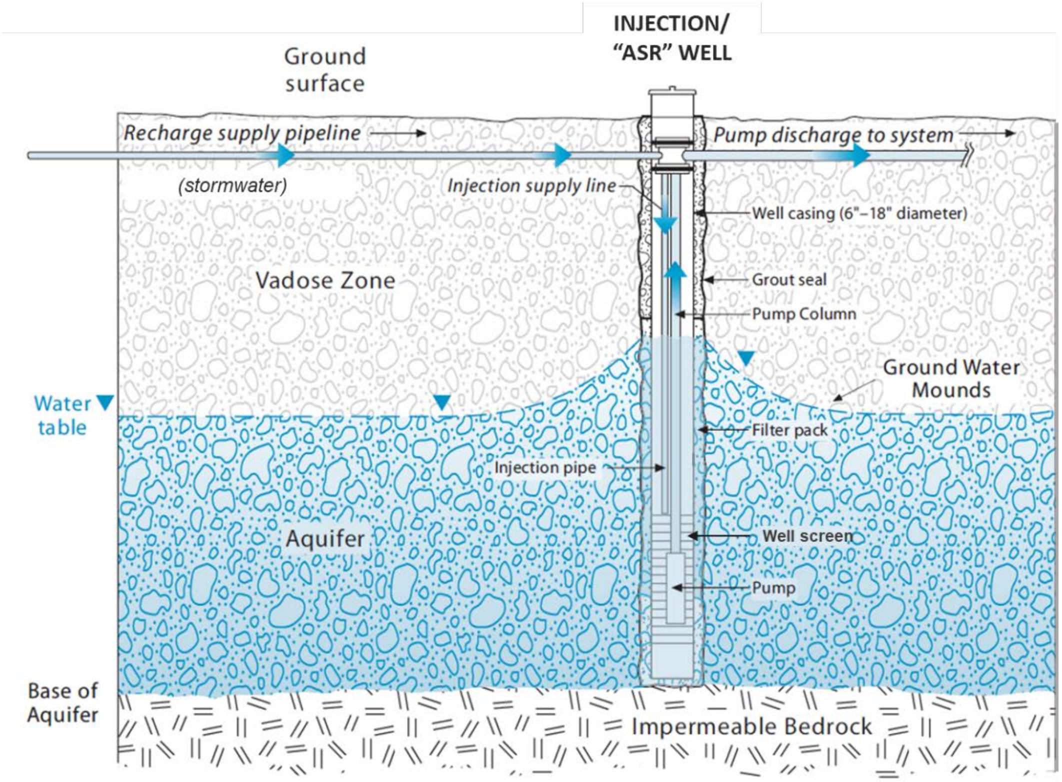

Figure 3-3. Examples of direct injection to an unconfined aquifer (adapted from Topper et al., 2004).

Stormwater can be injected directly into a saturated aquifer, raising the water table around the well. For ASR applications, the same well can later be used for recovery (Topper et al., 2004).

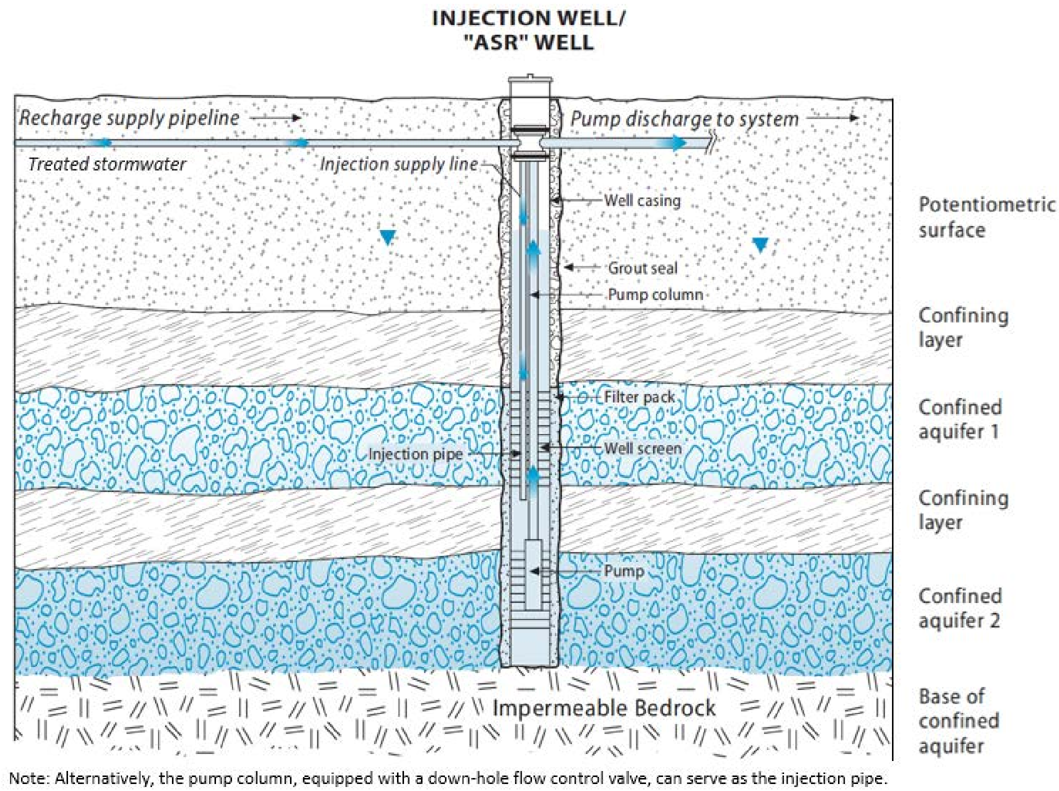

Figure 3-4. Example of direct injection in a confined aquifer (adapted from Topper et al., 2004). *Stormwater can be injected directly (using an injection pipe or a pump column with a down-hole flow control valve) into a confined aquifer. For ASR applications, the same well can later be used for recovery. Horizontal wells can be used for injection to increase the area of the well open to the aquifer (thus enabling greater injection rates) (Topper et al., 2004).*

4 HYDROGEOLOGY/WATER VOLUMES

The physical and hydroclimatic setting of an EAR system has a direct influence on its performance, including the volume of water the system can capture and recharge. This section discusses factors that influence the performance of EAR systems and provides examples of successful and unsuccessful systems to illustrate what is known and unknown about the effectiveness of stormwater EAR systems.

Table 4-1 lists the main factors shown to influence the performance of stormwater EAR systems. Note that none of these factors influence EAR independently; they all interact with each other. System design and geologic factors that also affect EAR system performance (with regard to flow regimes) are discussed elsewhere in this report. Performance of EAR systems with regard to pretreatment and protection of water quality is discussed in Section 6.

Table 4-1. Factors That Influence Stormwater Recharge Volumes

Factor	Mechanism	Relationship
Precipitation	Magnitude, frequency, duration, intensity, and seasonality	Increases in the magnitude, frequency, duration, and intensity of precipitation events generally increases runoff volumes, resulting in increased potential EAR volumes. However, if runoff volumes exceed EAR system design capacities, excess runoff may be lost to other surface water bodies. Similarly, seasonality of precipitation events affects EAR volume in that more intense storms may lead to more runoff and less recharge.
Evapotranspiration	Soil moisture deficit, surface evaporation, vegetative transpiration	As evapotranspiration increases, soil moisture deficit, surface evaporation, and vegetative transpiration will increase and EAR volume will decrease.
Land cover	Runoff generation (impervious surface), runoff reduction (pervious and vegetated surfaces), stormwater quality (developed areas)	More precipitation is converted to runoff when more area is covered by impervious surfaces. If that runoff can be captured by recharge systems, such as infiltration basins, then EAR volumes will increase. However, if the runoff intensity is too high, much of the runoff volume will bypass stormwater capture systems. Further, more impervious surface in developed areas generally results in degraded stormwater quality, which can increase clogging.
Climate change	Changing frequency, seasonality, and total precipitation, as well as changing temperatures	Increases in precipitation will generally result in increased potential for EAR. As noted above, more flashy, intense, or seasonal precipitation patterns can lead to exceedance of EAR system design, resulting in runoff losses to surface water bodies. When climate change leads to warming temperatures and increased evapotranspiration, EAR volumes will decrease.

Factor	Mechanism	Relationship
Soils	Soil type, grain size, hydraulic conductivity	Soil type and grain size affect the rate at which water can flow through a soil, which is generally expressed as hydraulic conductivity. Sandy soils generally have high conductivities and are conducive to sustained infiltration rates, whereas soils with high clays or fines will limit infiltration rates. Soil disturbance and compaction can also limit infiltration rates.
Geology	Clogging in fractured rock or fine-grained unconsolidated aquifers; dissolution of karstic substrate when exposed to mildly acidic stormwater	Clogging, as noted below, leads to decreased EAR volumes. Conversely, dissolution of karst can result in increased permeability at the recharge site, allowing for greater recharge rates and volumes.
Depth to water table	Higher groundwater tables lead to lower flow rates for infiltrating waters	EAR volumes decrease when the depth to the water table is too shallow.
Total suspended solids (TSS) in recharge water	Physical clogging	As the TSS concentration in the recharge water increases, physical clogging can increase, resulting in decreased EAR volume.
Nutrients and organic content in recharge water	Biological clogging	High nutrient and organic content (including organic contaminants) can promote biological growth, which can clog the system, resulting in decreased EAR volume.

4.1 STORMWATER AVAILABILITY

4.1.1 Precipitation

The characteristics of precipitation and runoff events, such as storm magnitude, frequency, duration, intensity, and seasonality, have a direct effect on the fraction of precipitation that is converted to recharge (sometimes referred to as the recharge to precipitation ratio, or RPR) in recharge systems. The RPR generally increases with increases in storm duration but deceases with increases in storm intensity or magnitude (Bhaskar et al., 2018; Tashie et al., 2016), reflecting the dynamic balance between system infiltration rates and the rate at which stormwater is delivered to a system.

Seasonal rainfall patterns are often a main driver of stormwater EAR adoption, as communities with long dry seasons often must look for ways to accumulate surplus water during the wet season. Although greater annual rainfall generally leads to greater potential for stormwater recharge, stormwater EAR can still be effective in areas with seasonal and low rainfall rates (Clark et al., 2015; Dahlke et al., 2018; Dillon et al., 2014; Milczarek et al., 2005; O’Leary et al., 2012; Racz et al., 2012). For example, many water-scarce locations receive their rainfall during a short wet-season, and it is not uncommon for most of the rainfall to come in a small number of large storm events. In California, distributed stormwater infiltration basins that capture runoff from basins on the order of 100–1,000 acres have proven effective in minimizing wet season runoff losses and recharging overdrawn aquifers (Beganskas, 2018; Beganskas and Fisher, 2017; O'Leary et al., 2012). Similarly, in South Australia, regional ASR systems using dry wells (Vanderzalm et al., 2014a, 2014b) or pretreatment systems, such as constructed wetlands, combined with injection wells (Clark et al., 2015; Dillon et al., 2014; Vanderzalm et al., 2014b) are becoming critical to maintaining municipal water supplies during the dry season.

Annual rainfall rates also influence municipal approaches to stormwater management (past and present) and potential for stormwater EAR adoption. Historically, nuisance flooding in wet and dry urban areas motivated diverse approaches to stormwater management with very different consequences. Many cities approached stormwater management by evacuating runoff from the landscape as rapidly as possible, leading to nutrient and other pollution problems in downstream waterbodies (Booth et al., 2016; Novotny, 1994). Luthy et al. (2019) describe problems in cities that are now trying to address both stormwater-based pollution problems and water scarcity, pointing to cities like Los Angeles, California (average annual rainfall of 15 inches/year) that are using distributed stormwater recharge practices to address both problems simultaneously (Hagekhalil et al., 2014; Sadeghi et al., 2017, 2018, 2019). In contrast to the problems created by Los Angeles's historic approach to stormwater management, the city of Chandler, Arizona (average annual rainfall of 8 inches/year), developed a system of recharge basins and dry wells to capture stormwater runoff instead of evacuating it from the landscape (Milczarek et al., 2005). The system has proven successful, avoiding all downstream water quality impacts by recharging all of the city's runoff within city limits—even with annual runoff volumes varying from 1,500 to 10,900 acre-feet per year between dry and wet seasons, respectively. This local recharge basin approach has also proven successful in wetter areas of the country. Beginning in the 1930s, Nassau County, New York (annual rainfall of 43 inches/year) began developing a system of more than 1,000 recharge basins to address roadway flooding concerns (Aronson and Prill, 1977; Bouwer et al., 2008; Weaver, 1971). Upon further urbanization, the primary role of the basins changed to stormwater recharge, raising local groundwater levels an estimated 5 feet above predevelopment levels (Bouwer et al., 2008). Note that Long Island's sandy, permeable soils may permit particularly rapid infiltration.

4.1.2 Evapotranspiration

"Evapotranspiration" refers to the combination of evaporation and transpiration processes. Stormwater that remains within the ponding area or vadose zone of an infiltration practice is subject to evaporation and, if the site is vegetated, transpiration. Vegetation's contribution to evapotranspiration, and its influence on recharge performance, can vary. In the arid southwest, vegetation likely increases evapotranspiration losses to the detriment of recharge volumes; in more humid climates like New York, the root zone's ability to break up the soil and increase recharge rates can more than compensate for its transpiration demand, leading to greater recharge performance (Bouwer et al., 2008). Evapotranspiration losses can be significant if the climate is dry (e.g., the arid southwest United States) or if infiltration rates are low. Losses from surface storage, which can be on the order of 30–50% (Arshad et al., 2014), are often a driver of subsurface storage considerations.

Evapotranspiration from stormwater infiltration practices is variable depending on system infiltration rates and local evaporative demand. In a 4-acre infiltration basin in the Pajaro Valley, California, evapotranspiration losses were less than 0.5% of the captured stormwater (Beganskas and Fisher, 2017). The authors attributed these low losses to reasonably high infiltration rates (averaged 0.3 meters/day over six years) and the timing of major rainfall events, which generally occurred during the cool and humid winter months when evaporative demand was low. Results from a nearby infiltration pond showed similarly low losses of less than 2% of total recharge volume (Racz et al., 2012). Conversely, in a study of four retention ponds in New Mexico that received drainage from 70 acres of developed land, Miller (2006) found that although the ponds captured 82% of the precipitation that fell on the drainage area, 40% of the captured volume was lost to evapotranspiration. These losses were largely attributed to the arid environment (annual precipitation of 9 inches/year), including a high evaporative demand and soil moisture deficits that develop between storm events.

4.1.3 Land Cover

Land cover influences stormwater runoff and recharge potential in a variety of ways. Impervious surfaces increase the amount of precipitation converted to stormwater runoff, which can increase the suitability of stormwater recharge systems. Pervious surfaces in both urban and rural areas tend to be vegetated and, depending on the soil infiltration rates, local climate evaporative demand and transpiration rate of the cover vegetation, recharge rates can be high or low depending on the balance between precipitation and evapotranspiration.

In the arid southwest, predevelopment recharge rates were often less than 1% of annual precipitation owing to rapid evapotranspiration of any precipitation by natural vegetation (Allison et al., 1994; Gee et al., 1994; Miller, 2006). However, urbanization in this region, which tends to trade vegetation for impervious surface, can increase runoff volumes which can then be routed to existing streams or stormwater basins, both of which tend to have higher infiltration rates than natural ground cover (Pool, 2005; Scanlon et al., 1999; Stephens et al., 2012). As an illustration of the unrealized potential for stormwater EAR in urban areas of the southwest, Green (2007) estimates that in the Los Angeles River Basin, urbanization and lined flood control channels have increased the amount of precipitation converted to runoff (and subsequently lost to the ocean) from 5% in 1920 to 50% in the 21st century. As discussed earlier, water managers in Los Angeles are starting to implement distributed stormwater practices to capture this water for EAR (Hagekhalil et al., 2014; Sadeghi et al., 2017, 2018, 2019).

Pervious land cover that is not associated with a recharge facility also promotes infiltration of stormwater, though recharge rates tend to be lower in such areas owing to less intense inflows of runoff. Still, when these lower recharge rates are spread over a larger area (as opposed to a smaller, more concentrated recharge practice), total recharge volumes can be comparable. In a study comparing the recharge rates and total recharge volumes between irrigated turf and an infiltration trench in San Francisco, CA, researchers found recharge rates to be lower over the turf but total recharge volume to be greater compared to an infiltration trench with a comparably sized drainage area (Newcomer et al., 2014). These results are, however, likely dependent on the high irrigation rate (approximately 32 inches/year, in addition to local rainfall of 28 inches/year) and the comparison of total recharge volumes would likely not apply for unirrigated pervious areas.

Land cover also influences sediment and pollutant transport, which can have a detrimental effect on infiltration systems. As shown in Table 4-1, poor water quality can lead to physical and biological clogging (Ashoori et al., 2019; Dallman and Spongberg, 2012; Maliva, 2020; Song et al., 2019). From the study of a four-acre infiltration basin receiving runoff from an active ranch containing large areas of unvegetated soil, Beganskas and Fisher (2017) found that although their study system included a sedimentation basin for pretreatment, the sedimentation basin tended to filter larger particles, leaving finer particles (e.g., silts and clays) to eventually accumulate in the infiltration basins. Still, given the ample time allowed for infiltration basin drawdown between storm events, infiltration rates were still high enough that stormwater recharge remained runoff limited rather than infiltration limited.

Infiltration systems located in urban areas must be able to mitigate the impacts of pollutants that can cause both physical and biological clogging. In South Australia, several stormwater ASR projects using injection wells have been implemented in developed catchments to varying success. The Parafield Airport (Clark et al., 2015; Dillon et al., 2014; Marchi et al., 2016) and Andrews Farm (Herczeg et al., 2004; Pavelic et al., 2006) ASR systems both receive runoff from mostly developed catchments (75% urban and 40% residential and industrial, respectively). They have multiple levels of pretreatment to address sediment (coarse and fine) and nutrients, including multiple settling basins and a wetland for nutrients in the case of the Parafield Airport site. Both sites have operated successfully for years, except for a brief clogging event at Andrews Farm due to a nutrient-enrichment-induced zooplankton bloom in one of the settling basins. This was remedied with a geotextile filter around the intake pump. In contrast, the Urrbrae Wetland ASR Project (Bouwer et al., 2008; Lin et al., 2006), a pilot system attempting to inject urban stormwater treated with a wetland and rapid sand filter, failed after just six weeks of operation. Failure was attributed to elevated amounts of leaf matter, organics, nutrients, and small amounts of motor oil that led to either physical clogging or biological clogging through biofilm formation on the well screens.

4.1.4 Climate Change

Many EAR systems function via physical and biological processes that are sensitive to changes in climate (e.g., changes in air temperature, precipitation). Consideration of climate change is thus relevant to planning for new EAR of stormwater. Climate change in the United States includes warming temperatures, and an increased frequency of heavy precipitation and runoff events (Reidmiller et al., 2018), which have consequences for stormwater (U.S. EPA, 2018a). Increases in the frequency, duration, and intensity of storm events can lead to problems such as nuisance flooding, as well as excessive stormwater flowing into pipes containing sewage and industrial wastewater in locations with combined sewer systems, leading to combined sewer overflows (Maxwell et al., 2018). Such changes could adversely affect stormwater quality. Conversely, some areas could experience reductions in precipitation, exacerbating existing water shortages (IPCC, 2008).

Changing precipitation patterns can influence the occurrence and practice of stormwater EAR, as drought will increase the need for conservation and how stormwater EAR systems function and are designed. For example, faced with prolonged droughts and more severe storms, Australia has embraced stormwater EAR as a means of addressing both problems simultaneously (Bekele et al., 2018; Clark et al., 2015; Dillon et al., 2014). Many areas of the United States are facing similar challenges. Based on an analysis of climate data from 1950 to 2009 for the 100 largest U.S. urban areas, Mishra and Lettenmaier (2011) found 30% of areas to have statistically significant changes in precipitation patterns, mostly in the form of increases in daily maximum intensities and number of days with heavy precipitation. This generally conforms with predictions of future precipitation patterns, which have many U.S. cities seeing more days with heavy precipitation (Maxwell et al., 2018). California, in particular, is expected to see decreases in total precipitation and precipitation event durations but increases in the frequency and magnitude of individual events (Dahlke et al., 2018; Dettinger, 2011; Pierce et al., 2013), both of which can lead to decreased groundwater recharge as more rainfall is converted to runoff. This conforms with the findings of Newcomer et al. (2014), who simulated the performance of a suburban infiltration trench under future climate scenarios and found current design standards to be inadequate for capture of future runoff volumes.

Conversely, there are areas of the country where precipitation events are expected to become less frequent. Combined with higher temperatures, this is likely to lead to more frequent and prolonged drought conditions (Lall et al., 2018). Even on a shorter timescale, multidecadal climate variability patterns, such as the El Niño/Southern Oscillation, can increase the intensity of both storm and drought conditions (Ropelewski and Halpert, 1986). Because EAR can help reduce flood impacts in areas with more frequent precipitation and maintain groundwater recharge rates in areas with less frequent precipitation, it is well-suited to short- and long-term climate change adaptation.

Climate change effects on EAR will also interact with land cover (Clark et al., 2015; Dillon et al., 2014; Pool, 2005). In an analysis of a 3,930-acre catchment that captures stormwater for water supply augmentation, Clark et al. (2015) found that the reductions in stormwater runoff anticipated due to a drying climate were less pronounced within their urban study system compared to the surrounding rural (and largely pervious) areas. This was attributed to the drying climate increasing the soil moisture deficit of pervious areas, which absorb a larger portion of rainfall in rural areas compared to urban areas.

4.2 SITE CHARACTERISTICS

4.2.1 Soils

Suitable locations for stormwater EAR require permeable soils that are conducive to sustained infiltration (Bouwer, 2002; Maliva, 2020). Some infrastructure, such as dry wells and other injection wells, can circumvent unsuitable surface soils with sufficiently deep wells, though *sufficient* in this case is highly site-specific: depths range from less than 10 feet for some dry wells (Geosyntec, 2020; Talebi and Pitt, 2014) to hundreds of feet for injection wells (Page et al., 2011).

A relatively small difference in the percentage of fine material—e.g., silts and clays—can have a large influence on the saturated hydraulic conductivity and overall infiltration rates of surface practices such as infiltration basins (Racz et al., 2012). Infiltration rates may also be subject to large seasonal and temperature variations (Constantz et al., 1994; Emerson and Traver, 2008; Jaynes, 1990; Ronan et al., 1998; Schuh, 1990). Site characterization is critical to determining the performance of infiltration systems. For example, estimates of expected soil infiltration rates are used to calculate an area of infiltration basin needed to meet target recharge rates (Bouwer, 2002; Bouwer et al., 2008; Maliva, 2020).

In urban areas, where surficial soils tend to be disturbed or compacted from development activities but well-draining subsurface soils may still be intact, it is also important to characterize soils vertically. Talebi and Pitt (2014), in an investigation of dry well infiltration rates in New Jersey, note that for sites with poorly infiltrating surficial soils (e.g., Hydrologic Soils Groups C and D), even shallow dry wells (on the order of several feet deep) can penetrate deep enough to access better draining soils and maintain sufficiently high infiltration rates. They also note that general National Resources Conservation Service soil maps may not include accurate characterization of subsurface soils, particularly in urban areas. These maps were developed for agricultural purposes and have been used for many applications beyond what they were intended for. Given that most of the surveys associated with these maps end at one-meter depth at most, it is important to take great care when using them in the context of EAR projects, which will interact with much deeper subsurface geology.

4.2.2 Geology

Geological conditions have a large influence on the fate of infiltrated stormwater—but characterizing subsurface heterogeneity is challenging. Aquifer porosities, which have a large influence on aquifer storage capacity and hydraulic conductivity, vary widely by type (Maliva, 2020). Intergranular dominated aquifers have porosities in the range of 10–45% and fractured rock aquifers often have porosities of less than 1%. Solution conduit aquifers, of which karstic aquifers are a common type, typically have low overall porosities but significant hydraulic connectivity via the conduits.

For injection wells, limestone or karstic aquifers generally maintain higher permeability than unconsolidated, intergranular aquifers due to the larger pore spaces and anti-clogging effect of matrix dissolution (Bouwer et al., 2008; Dillon and Pavelic, 1996; Maliva et al., 2020). If stormwater is lower in pH than the aquifer, the stormwater can dissolve some of the clogging that that might occur due to sediment or biological activity (Bouwer et al., 2008). At the Andrews Farm ASR site in South Australia, well clogging due to prolonged injection of turbid, urban-derived stormwater was partially offset by calcite dissolution of the aquifer matrix and routine well redevelopment (Pavelic et al., 2006). Conversely, if stormwater has a high mineral content, mineral precipitation can clog soil or aquifer pores (Maliva, 2020).

In addition to direct characterization of geological conditions, predevelopment hydrologic patterns can be a good indicator of whether local geological conditions are amenable to stormwater recharge. The town of Mount Gambier, Australia, is based around Blue Lake, which has served as the town’s drinking water supply for over 140 years (Dillon et al., 2014). Natural sinkholes originally provided a direct link between stormwater and the lake. As the town developed, sinkholes were supplemented with dry wells, maintaining the connection between stormwater and lake replenishment to the extent that the estimated annual recharge volume is roughly equivalent to the volume of water pumped from the lake for water supply each year (Dillon et al., 2014; Vanderzalm et al., 2014a).

In southeastern Florida, urban development in Miami, along the eastern edge of Everglades National Park, has resulted in increased surface runoff and reduced groundwater recharge. This has lowered the water table underneath Miami over time, causing an increase in groundwater export from the adjacent park. The C-111 project, consisting of a series of infiltration basins overlying the naturally porous karstic substrate, was implemented to create a groundwater mound, or buffer, between the park and Miami and curtail shallow groundwater losses (Bouwer et al., 2008; Brown et al., 2014). Likely owing to the karst geology of the area, the system has shown little to no sign of clogging, despite having no pretreatment and receiving high TSS runoff during the wet season (Bouwer et al., 2008).

Miami is also one of many North American coastal communities experiencing the effects of saltwater intrusion by virtue of its local hydrogeological conditions and historical groundwater withdrawal rates that have exceeded recharge rates (Barlow and Reichard, 2010; Prinos et al., 2014). EAR can help to mitigate such problems by restoring groundwater levels such that flow directions are reversed. Still, geologic complexities make it difficult to determine the extent of impacts and the effectiveness of mitigation strategies. Costall et al. (2020) combined numerical simulation, geophysics, and analysis of more than 30 years of data at one site to conclude that determining the landward extent of the seawater interface is extremely challenging. Heterogeneity in aquifer parameters has a large influence on migration of this interface. Saltwater intrusion is discussed further in Section 6.5.

4.2.3 Water Table Depth

Water table depth can be an important factor that affects the performance of certain stormwater EAR practices. For systems, such as infiltration basins and much green infrastructure, that recharge directly into surficial, unconfined aquifers, high water tables that are close to or even intersect with the bottom of the infiltration system can severely limit infiltration rates (Talebi and Pitt, 2014; Petrides et al., 2015). For systems that are designed to bypass surficial storage zones and recharge deeper, confined aquifers (which is often the case for injection wells), high water tables may be less of a concern.

Design guidelines for many practices used in stormwater EAR suggest (and regulations often require) a minimum distance between the bottom of an infiltration system and the seasonal high groundwater table. The minimum has been set at 2–3 feet for practices such as dry wells or infiltration basins (e.g., NJDEP, 2021) and can be as much as 10 feet in some locations (e.g., Geosyntec, 2020). The reason is that, generally, once the groundwater table (or capillary fringe) intersects the bottom of the infiltration system due to short-term mounding, the infiltration pathway shifts from a downward flux through the unsaturated zone to a lateral flux out of the perimeter of the system (Bouwer, 2002; Petrides et al., 2015). This can significantly reduce overall drainage rates, as shown through extensive physical modeling and field observations (Bhaskar et al., 2018; Bouwer, 2002; Talebi and Pitt, 2014; Petrides et al., 2015). Therefore, most designs incorporate a minimum distance as a safety factor.

Water table depth can be highly variable, both spatially and temporally. The benefits of EAR may also be greater in arid and semiarid regions where depths to water are great. In areas with high water tables, benefits of EAR may be offset by potential damages to basements, foundations, and subsurface transportation infrastructure if water tables rise too high.

4.3 PERFORMANCE OF EAR SYSTEMS

Table 4-2 summarizes the attributes of the stormwater EAR case studies identified in this review. The table divides the systems studied into two broad categories according to whether infiltration is passive or active. (Infiltration basins, infiltration trenches, green infrastructure, and dry wells are passive systems;

injection well systems are active systems.) It also provides system size, location, estimated recharge rates, annual recharge volumes, and other notable attributes.

4.3.1 Infiltration Rate

In case studies that include infiltration rates, estimates range from about 0.1 to 10 feet/day. Of those studies that noted infiltration rates less than 1 foot/day, one was of a small, on-lot infiltration trench with the infiltration rate calculated as a theoretical draw-down of the storage volume over 24 hours per design requirements (Newcomer et al., 2014); the actual infiltration rate was likely higher, as drawdowns generally took less than 24 hours. Two other studies, both of regional infiltration basins in the Pajaro Valley of California, noted infiltration rates less than 1 foot/day, but these low rates were observed at the end of the wet season and attributed to the seasonal accumulation of sediment or a drop in basin stages (Beganskas and Fisher, 2017; Racz et al., 2012). A maximum infiltration rate of up to 135 feet/day was noted by Bouwer et al. (2008) in a system of infiltration basins in Nassau County, New York. Generally, of the examples included in Table 4-2, most EAR volumes were limited by water (i.e., runoff) availability rather than infiltration rates (Aronson and Prill, 1977; Beganskas and Fisher, 2017; Milczarek et al., 2005; Miller, 2006).

4.3.2 Injection Rate

Many injection well systems rely on active pumping to maintain a design injection rate. To buffer the intermittent nature of storm events and use injection pumps more consistently and cost-effectively, injection well systems generally include upstream storage basins that often double as pretreatment (Dillon and Pavelic, 1996; Dillon et al., 2014; Page et al., 2011; Vanderzalm et al., 2014b). Injection rates are therefore generally a function of the number and size of injection pumps, which are designed according to the volume of stormwater that can be captured and temporarily stored. Of the injection well studies in Table 4-2, injection rates ranged from 0.012 to 3.2 cfs. The Parafield Airport (Australia) site, one of the most well-described systems, uses a wetland to treat runoff from a 3,930-acre residential and industrial catchment. Total storage capacity is 26 million gallons, and total injection rate is 3.2 cfs, or 2 MGD, using four wells (Clark et al., 2015; Dillon et al., 2014; Marchi et al., 2016; Vanderzalm et al., 2014b). The system is large enough to capture and inject 80% of annual runoff (15% of precipitation over the contributing area); on recovery, this is enough to reliably meet the town’s daily demand of 0.37–0.62 MGD, the majority of which is used for seasonal irrigation (Clark et al., 2015). Some injection wells also use gravity as the driving force when injection is into an unconfined or low-pressure aquifer (Pyne, 1997, 2005).

4.3.3 Recharge Efficiency

As detailed above, measures of recharge effectiveness or system efficiency are confounded by numerous factors and even more ways in which those factors interact. Still, at a basic level, *recharge efficiency*—defined here as the percentage of rainfall over a system’s drainage area that contributes to recharge—is one measure of efficiency that can apply to any system. Note that this is slightly different from the definition adopted by Newcomer et al. (2014) of percent of system inflow that contributes to recharge. In the studies that either note recharge efficiency or provide enough information to calculate it, values range from 6% to 60%, with most falling below 30% (Beganskas and Fisher, 2017; Clark et al., 2015; Milczarek et al., 2005; Miller, 2006; Newcomer et al., 2014; Sadeghi et al., 2017; Vanderzalm et al., 2014a, 2014b); this does not include the Andrews Farm pilot study, which was not designed to maximize recharge (Herczeg et al., 2004; Pavelic et al., 2006). Recharge efficiency is not necessarily correlated with annual precipitation: the highest value, 60%, was reported for a system receiving an average of 9 inches/year (Miller, 2006).

Although 30% may not seem high, the recharge efficiencies of stormwater recharge systems can be substantial compared to predevelopment recharge rates owing to the dynamic balance between precipitation, evapotranspiration, and land cover discussed above. In arid environments, natural recharge can be less than 1% of precipitation (Allison et al., 1994; Gee et al., 1994). Yet, especially in arid environments, recharge efficiencies can be an order of magnitude greater (Milczarek et al., 2005; Miller, 2006). In urban environments, impervious surfaces reduce natural recharge rates, and the volume of runoff lost to downstream waters (leading to water quality impairment) can be of the same magnitude as the volume captured by distributed stormwater infiltration systems (Sadeghi et al., 2017). Even compared to turfgrass, which allows for a certain level of infiltration, recharge rates of infiltration systems were found to be 2–3 times greater owing to more concentrated inflows and less evapotranspiration loss (Newcomer et al., 2014).

Table 4-2. Summary of Stormwater EAR Case Studies

Type of EAR System	Location	Precip-itation (in/yr)[1]	Recharge Volume (ac-ft/yr)[2]	Recharge Rate[2]	Recharge Efficiency[3]	Citation	Comments
Infiltration basin	Pajaro Valley, CA	22	71	Average: 1ft/d Max when basin filled:3–9 ft/d	27%	Beganskas and Fisher, 2017	• Six-year study of distributed MAR • Pretreatment for large solids • 172-acre contributing area • Despite heavy sedimentation, system remained runoff-limited rather than infiltration-limited
Infiltration basin	Pajaro Valley, CA	20	Permitted up to 2,000	0.3–4 ft/d	NA	Racz et al., 2012	• More than 90% of precipitation falls from December through April • Runoff from slough treated with a sand pack filter, pumped to a recharge pond in a natural depression
Infiltration basin	Stockton, CA	14	5,070	NA	NA	O'Leary et al., 2012	• Basin receives stormwater in the wet season, surface water diversions during the dry season • System undergoes four to nine recharge cycles per wet season (average 4,400 acre-feet per year) • Average of 670 acre-feet per year infiltrated during dry season from surface water diversions

Type of EAR System	Location	Precip-itation (in/yr)[1]	Recharge Volume (ac-ft/yr)[2]	Recharge Rate[2]	Recharge Efficiency[3]	Citation	Comments
Infiltration basins	Nassau County, NY	43	68,000	43.9 ft/d (3.12–135 ft/d)	NA	Aronson and Prill, 1977; Bouwer et al., 2008	• About 2,200 basins with an average surface area of 1.5 acres, most constructed in the 1930s • Average recharge of 21 feet/year/basin • Over time, major reasons for failure (clogging) include substrate permeability, drainage area land use, basin age, and intersection of basin floor with the water table
Infiltration basins	New Mexico	9	38	NA	60%	Miller, 2006	• Old mine site with commercial buildings and roadways, 70-acre contributing area to four earthen retention ponds • 82% of precipitation (13 inches/year) makes it to the ponds, 60% of precipitation is recharged • Recharge is runoff-limited
Infiltration basins	Homestead, FL	61	142,000	2 ft/d	NA	Bouwer et al., 2008	• Source of recharge water is wet weather overflows from the C-111 canal, mostly due to wet season stormwater • Annual recharge rate representative of 2004 total • Minimal clogging despite absence of pretreatment or settling basins

Type of EAR System	Location	Precip-itation (in/yr)[1]	Recharge Volume (ac-ft/yr)[2]	Recharge Rate[2]	Recharge Efficiency[3]	Citation	Comments
Infiltration trench	San Francisco, CA	28	0.015–0.033	0.6 ft/d	6–13%	Newcomer et al., 2014	• Drainage area of 0.11 acres • Recharge rate calculated assuming volume (71 cubic feet) of the 118-square-foot trench drains in 24 hours • Recharge of 58–79% of inflow
Stormwater control measures (see comments)	Los Angeles, CA	15	41	NA	15%	Sadeghi et al., 2017	• More than 30 distributed low-impact development practices treating 220 acres of urban area • Provides 41 acre-feet of infiltration per year • A combination of rain gardens, infiltration trenches, large dry wells, and a large infiltration gallery underneath a parking lot
Infiltration basins and dry wells	Chandler, AZ	8.3	770–8,700	> 1 ft/d	6–18%	Milczarek et al., 2005	• Analysis of dry, wet, and normal year water balance • Systems have a maximum ponding depth of 3 feet and must drain within 36 hours, so minimum recharge rate is 1 foot/day
Dry wells	Millburn Township, NJ	49	NA	< 1 to > 6 ft/d	NA	Talebi and Pitt, 2014	• Wells are typically ~10-foot-diameter cylinders, 6 feet deep with 2 feet of soil overtop and 2 feet of gravel underneath • Infiltration rates found to depend on groundwater table depth and subsurface soil conditions

Type of EAR System	Location	Precip-itation (in/yr)[1]	Recharge Volume (ac-ft/yr)[2]	Recharge Rate[2]	Recharge Efficiency[3]	Citation	Comments
Drainage wells	Mount Gambier, South Australia	28	4,100	NA	27%	Vanderzalm et al., 2014a, 2014b	• Drainage area of about 6,500 acres (average 14 acres/well), 41% impervious surface • Direct connection between stormwater drainage and lake, which has served as the potable supply for the city since the late 1880s
ASR injection well	Aspendale, Australia	29	0.67–2.0 per cycle	0.012–0.014 cfs	NA	Page et al., 2011	• Field trial to evaluate alternative stormwater pretreatment options • One injection well and one recovery well • Injecting stormwater dropped injection rate by about 20% compared to potable
ASR injection well	Adelaide, Australia (Urrbrae Wetland ASR Project)	18	NA (early failure due to clogging)	0.078 cfs	NA	Bouwer et al., 2008; Lin et al., 2006	• Pilot study: 8-inch PVC injection well into unconsolidated siliceous aquifer • Source of recharge water was wetland-treated urban stormwater with 1 mm rapid sand filter pretreatment • Failed after six weeks of operation at 0.078 cfs (35 gpm) • Failure attributed to poor removal of suspended material, colloidal material, and elevated total organic carbon by the wetland and sand filter

Type of EAR System	Location	Precip-itation (in/yr)[1]	Recharge Volume (ac-ft/yr)[2]	Recharge Rate[2]	Recharge Efficiency[3]	Citation	Comments
ASR injection well	Adelaide, Australia (Parafield Airport Site)	18	892	3.2 cfs	15%	Clark et al., 2015; Dillon et al., 2014; Marchi et al., 2016; Vanderzalm et al., 2014b	• Injection of reedbed effluent into limestone aquifer • 3,930-acre residential and industrial catchment, 40% impervious surface area • Storage capacity of 26 million gallons • 4 wells with an injection capacity of 3.2 cfs • Runoff is 17.6% of rainfall, recharge is 14.6% of rainfall (18 inches/year)
ASR injection well	Andrews Farm, South Australia	23	52	0.53–0.71 cfs	0.2%	Herczeg et al., 2004; Pavelic et al., 2006	• Five-year ASR pilot study • Injectate from three interconnected detention basins receiving drainage from a 13,500-acre catchment • Injection rates maintained through periodic redevelopment (airlifting) despite high inflow TSS concentrations

1 U.S. data obtained from Fick and Hijmans (2017) when not available in the cited literature.

2 Average values estimated where ranges were reported.

3 Defined here as the percentage of rainfall over a system’s drainage area that contributes to recharge.

5 WATER QUALITY

One of the greatest concerns about the effects of EAR on groundwater is that contaminated stormwater, especially in the absence of the pretreatment systems typical in aquifer recharge operations that use wastewater, can pose risks to groundwater quality. Urban stormwater can contain a wide range of contaminants including nutrients, pathogens, metals, organics, and a range of emerging contaminants (Masoner et al., 2019; NASEM, 2016; Pitt et al., 1994, 1995, 2015; U.S. EPA, 1983). Currently, the National Stormwater Quality Database (NSDQ; https://www.bmpdatabase.org/nsqd.html) (Pitt et al., 2015) provides data on urban stormwater quality; it has been maintained since 2001 by the University of Alabama with support from EPA (Pitt et al., 2015). There is limited monitoring/sampling of stormwater in most urban areas; moreover, the specific source areas of contaminants in stormwater are not well understood. This is in part because mixed use zoning makes identification of stormwater contamination sources challenging (NASEM, 2016). The concentrations of contaminants in stormwater are also variable in time, seasonally, and during events with the "first flush," or first pulse of runoff, generally having much greater concentrations of those contaminants that had accumulated in the watershed between storm events. This section discusses the risks of EAR using stormwater, in particular the risks of water quality degradation.

In urban environments, the characteristics of materials used in building and landscaping can be more important in determining stormwater quality than the land use itself. For example, galvanized roofs are often associated with elevated zinc concentrations (NASEM, 2016; Pitt et al., 1994, 1995), old and decaying paint with elevated lead concentrations (Bannerman et al., 1996), and fertilized landscaping with elevated nitrogen and phosphorus concentrations (NASEM, 2016).

Other direct relationships between land use and contaminant loads in stormwater are known, such as the influence of roads on metals loads and the influence of direct runoff from fueling stations and locations of current or former underground storage tanks (USTs) on the presence of methyl tert-butyl ether (MTBE) and other hydrocarbons in stormwater (Borden et al., 2002). For example, a new industrial area with updated building materials may not introduce metals such as zinc, copper, and lead to stormwater at the same rate as older industrial areas, while residential areas with aging roof runoff capture systems could have an outsized effect on the presence of metals in stormwater (NASEM, 2016). Urban areas with poorly operating sewer systems and septic systems, for example those in older residential areas, are also associated with detection of pathogens, primarily fecal coliforms, in stormwater.

In areas with snow and ice, runoff generated from impervious surfaces often contributes to high salt concentrations in stormwater during the winter and early spring. The high rates of salt application for deicing of freeways, streets, and parking lots are likely sources of salt in stormwater. While industrial and commercial areas are often associated with high levels of metals in stormwater due to roofing materials and roof runoff capture systems, residential and commercial areas are often associated with abundant landscaping that contributes to the presence of organic chemicals such as insecticides, herbicides, and fungicides (associated with lawn and garden care) in stormwater.

Certain site geologies and practices can also create relatively direct pathways between runoff and aquifers and can therefore require additional screening or protection measures to reduce the potential for water quality impacts. Karst geologies—carbonate-based substrates typified by cavernous formations and sinkholes—can represent a direct conduit between the land surface and aquifers and often require special siting, design, and construction protections (e.g., CSN, 2009). Dry wells, which are engineered practices but can represent a similarly direct conduit to aquifers, often require special land use risk characterizations or screening procedures by design engineers or planners before construction (Geosyntec, 2020). Additional information on drywell best practices is included in Section 6.7.

Identifying source areas for organic contaminants in urban stormwater is challenging. Automobile exhaust and paving materials are known sources of polyaromatic hydrocarbons (PAHs), so by extension urban areas with abundant paved surface and high rates of automobile use, such as transit depots, could be potential source areas for organic compounds in stormwater. Similarly, industrial manufacturing processes and storage of consumer goods are known sources of organic contaminants, although their direct effect on stormwater contamination is not well characterized (NASEM, 2016).

As there is little to no sampling of aquifers adjacent to infiltration sites, this section focuses on literature regarding urban stormwater quality in general. However, additional work is needed on assessing urban stormwater quality adjacent to infiltration sites and the potential impacts to groundwater quality.

5.1 WATER QUALITY—PATHOGENS IN STORMWATER

5.1.1 Pathogen Occurrence in Stormwater and at EAR Sites

Pathogens can enter stormwater through a variety of sources, including both point and nonpoint sources (Ahmed et al., 2019). Although some information is available on the occurrence of pathogens in stormwater, the National Academies of Sciences and Medicine have identified research on the occurrence and fate of pathogens in stormwater as a research need (NASEM, 2016). Human pathogens generally enter stormwater through sewer overflows and leakages (NASEM, 2016; Page et al., 2016a; Pitt et al., 2003). Nevertheless, pathogens can also enter urban and suburban stormwater through defective septic systems, sewer overflows, and defecation from wild and domestic animals (Ahmed et al., 2019; NASEM, 2016). Although the profile of pathogens in stormwater shares some characteristics with that of wastewater, stormwater generally contains different types and concentrations of pathogens than wastewater (Ahmed et al., 2019).

Although the universe of waterborne pathogens includes many types of bacteria, viruses, and protozoa, only a few types of pathogens in stormwater are generally monitored for research studies or regulatory purposes. Fecal indicator bacteria (FIB) are monitored and serve as indicators of the presence of multiple other pathogens (ASCE, 2014; NASEM, 2016). This is a common approach in stormwater monitoring; for stormwater, FIB serve as indicators for the presence of pathogenic strains of fecal coliform (including *E. coli*), as well as streptococci and enterococci (NASEM, 2016; U.S. EPA, 2015). These FIB are sometimes measured as part of EAR projects (Asaf et al., 2004; Frias, R. III et al., 2008; Stone et al., 2019). The pathogens of greatest concern in groundwater are enteroviruses, *Shigella*, *Pseudomonas aeruginosa*, and various protozoa, such as *Cryptosporidium* (Clark and Pitt, 2007; Pitt et al., 1999, 2003).

While FIB monitoring may not be a perfect reflection of human health risks (Ahmed et al., 2019; NASEM, 2016), EPA considers FIB to be the best indicators of human health risks from pathogens in stormwater. EPA also accepts alternative indicators, such as *Bacteroidales*, *Clostridium perfringens*, human enteric viruses, and coliphages (ASCE, 2014). Generally, though, monitoring of stormwater focuses on quantifying *E. coli* and *Enterococcus* species (Ahmed et al., 2019).

When indicators are not used, researchers generally sample for the following pathogens (NASEM, 2016):

- Enteroviruses may also be chosen to represent enteric viruses.
- Rotaviruses may be chosen to represent “worst-case” scenarios for viruses.
- *Cryptosporidium* and *Salmonella* are also commonly measured.

While important for the design of EAR systems, data on pathogen concentrations in stormwater are limited in part because it is challenging to collect stormwater samples during storms. Some studies collect grab samples to characterize pathogen or FIB concentrations. However, automated samplers provide data that are more accurate and appropriate than grab samples. Automated samplers need to be installed and require construction of infrastructure and regular maintenance (Ahmed et al., 2019).

Still, some studies provide reference values for pathogens in stormwater. Page et al. (2016b) document that the 95th percentile reference values for adenoviruses, *Cryptosporidium*, and *Campylobacter* in urban stormwater are 2 numbers per liter (n/L), 1.4 n/L, and 11 n/L, respectively. Ahmed et al. (2019) also provide a summary of concentrations of various pathogens in stormwater compiled from the literature, showing most stormwater samples exceed the threshold values for recreational use in Australia. An EPA report (U.S. EPA, 2015) indicates that typical fecal coliform concentrations in urban runoff range from 400 to 50,000 n/100 mL. An additional source of data (fecal coliform, fecal streptococci, total coliform, and *E. coli*) is the NSQD (Pitt et al., 2015), which contains monitoring data from NPDES (National Pollutant Discharge Elimination System) MS4 (municipal separate storm sewer system) stormwater permit holders along with other sources such as the U.S. Geological Survey (USGS), academic research projects, and the Nationwide Urban Nationwide Urban Runoff Program (NURP) study (U.S. EPA, 1983).

Available data suggest the following observations on the siting and design of EAR systems:

- **Land use type:** Urban and high-density areas have higher *E. coli* concentrations than lightly or sparsely populated residential areas (Ahmed et al., 2019).
- **Season:** Concentrations of pathogens in stormwater may be much lower in the winter than in the summer (ASCE, 2014; Selvakumar and Borst, 2006). Some studies have found pathogen concentrations up to 20 times greater in warm months than in colder months (U.S. EPA, 2015). However, other researchers have noted that microorganisms can survive in the subsurface for months to years at temperatures below 4°C, whereas inactivation occurs more rapidly at higher temperatures (Pitt et al., 1999; Shen et al., 2020).
- **Precipitation and climate:** Fecal coliform loading from nonpoint sources is expected to increase with wetter and warmer futures. On the other hand, warmer and drier futures may reduce fecal coliform loading from nonpoint sources (Coffey et al., 2020).

In additional to stormwater, EAR operations themselves may be a source of pathogens at some sites. While pretreatment and aquifer treatment may remove pathogens from stormwater, stormwater EAR operations can in some cases promote the growth of pathogens. For example, stormwater ponds are at risk for cyanobacteria contamination. These bacteria release cyanotoxins that present public health concerns (O'Reilly et al., 2011). However, based on laboratory studies, sorption media specifically for cyanotoxins may remove such risks in stormwater ponds (O'Reilly et al., 2011). Any BMP with standing water can be a breeding ground for other bacteria as well (U.S. EPA, 2015).

5.1.2 Pathogen Fate and Transport

Maliva (2020) and Zhang et al. (2013) show that pathogen concentrations in groundwater are reduced by:

- Physical retention (filtration, straining, sedimentation, and adhesion)
- Inactivation (dying off)
- Dilution

While some attenuation of pathogens can be expected within an aquifer, physical retention rates tend to vary widely (de Lambert et al., 2021; Maliva, 2020; U.S. EPA, 2018b; Zhang et al., 2013). Physical retention rates within an aquifer depend on soil type, solute characteristics, dissolved organic matter, infiltration rate, rainfall frequency, rainfall intensity, and type of organism (de Lambert et al., 2021; Maliva, 2020; Zhang et al., 2013). Attenuation of pathogens is dependent on the following factors:

- **Type of pathogen:** While bacteria tend to rapidly die off, viruses and protozoa generally persist for longer in groundwater (U.S. EPA, 2018b). Bacteria and protozoa tend to die off more than 100 times faster and five to 10 times faster than viruses, respectively.
- **Population of existing microorganisms within the aquifer:** This can create competition for the existing resources within the aquifer and can also sometimes lead to predation upon introduced pathogens (U.S. EPA, 2018b).
- **Soil moisture:** The higher the soil moisture, the longer microorganisms generally can persist (U.S. EPA, 2018b).
- **Temperature:** As temperature increases, inactivation rates increase (U.S. EPA, 2018b).
- **Groundwater chemistry:** Retention of pathogens has been found to depend on salinity, redox state, dissolved oxygen concentration, and nutrient concentrations (Sidhu et al., 2015; U.S. EPA, 2018b; Zhang et al., 2013).
- **Travel time:** Unless they encounter conditions suitable for growth, all pathogens have non-zero die-off rates, meaning that, generally, longer travel times lead to lower pathogen numbers. Infiltration sites with a protective layer of surficial soils are therefore less vulnerable to contamination (de Lambert et al., 2021)

In a review of several case studies of EAR using stormwater or reclaimed wastewater, Page et al. (2010a) found that aquifers can play a significant role in removing pathogens from such waters and thus reducing human health risks. As noted above, bacteria generally decay the fastest of all pathogens, protozoa (e.g., *Cryptosporidium*) decay more slowly, and enteric viruses decay even more slowly (Pitt et al., 1999; Sidhu et al., 2015). Similarly, rotavirus and *Cryptosporidium* have been found to have variable removal rates, through inactivation and attenuation, in aquifers (Page et al., 2010a). Other studies have acknowledged that health-based targets for groundwater are hardest to achieve for viruses. If health-based targets for viruses have been achieved, it is likely that targets for protozoa and bacteria have also been achieved, although this should be verified (Page et al., 2012).

Through inactivation and attenuation, water drawn from the subsurface may have pathogen concentrations lower than concentrations in recharged stormwater. The subsurface may inactivate pathogens through a variety of processes, including die-off and retention (Pitt et al., 1999). Site-specific geochemical factors will affect pathogen survival in the aquifer (Sidhu et al., 2015) and attenuation in the aquifer (Clark and Pitt, 2007). For example, cyclical aerobic-anaerobic conditions in the vadose zone may cause some pathogens to die (ASCE, 2014). Increasing the vadose zone thickness may inactivate pathogens (Voisin et al., 2018). However, the subsurface is conducive to microorganism growth in one way: moving deeper into the subsurface can protect microorganisms because there is no UV radiation (ASCE, 2014).

Ambient groundwater will generally have lower pathogen concentrations than recharged stormwater. However, in column studies, Sasidharan et al. (2017) found that aquifer sediment removed more than 92.3% of viruses under all considered EAR conditions, indicating the potential for low pathogen levels reaching groundwater or aquifers. Yet some researchers caution against using laboratory studies for estimates of pathogen behavior in the field (Page et al., 2015a; Sidhu and Toze, 2012). Laboratory studies cannot properly replicate aquifer conditions (Page et al., 2015a) and may underestimate the survival potential of pathogens, enteric viruses in particular (Sidhu and Toze, 2012).

In one study of an EAR system, groundwater with a residence time of greater than 14 days was found to remove all viruses, through attenuation and die-off, to concentrations below detection limits (Betancourt et al., 2014). Long residence times are necessary for ensuring pathogen die-off (Bekele et al., 2014). In another study, bacteria had a one $\log_{10}$ reduction inactivation time of less than 2.5 days in an aquifer (Sidhu et al., 2010). Similarly, field studies in Australia have indicated that *E. coli* concentrations in recovered water were 90–99% lower than in injected stormwater water because of inactivation and attenuation (Page et al., 2015a). The results of the summarized studies above may not be applicable to all EAR sites, as pathogen inactivation and attenuation are dependent on site-specific characteristics.

While there is limited information on attenuation of pathogens in the subsurface in stormwater EAR operations in the United States, field work has been done in Australia to characterize removal of pathogens over time (Page et al., 2010a, 2012, 2016b). For example, the Parafield Gardens site in the city of Salisbury has been harvesting urban stormwater from a mixed industrial and residential neighborhood and passing it through constructed wetlands, ASR systems, and aquifer storage transfer and recovery systems (Clark et al., 2015; Dillon et al., 2014; Page et al., 2010b). The system in Salisbury was set up with two stormwater settling basins and an engineered wetland through which the stormwater passes before injection. The injected stormwater is recovered via separate wells after a minimum travel distance of 50 meters and a mean aquifer residence time of nine months (Page et al., 2010c). Rotavirus, *Cryptosporidium*, and *Campylobacter* were used as representative pathogens for viruses, protozoa, and bacteria, respectively. Table 5-1 below presents the removal efficiencies of each of these pathogens, in $\log_{10}$ removal (Page et al., 2010c).

Table 5-1. Comparison of Aquifer Log_{10} Removal of Pathogens to Other Treatment Technologies

	Aquifer Removal			Secondary Treatment		UV Disinfection			Reverse Osmosis	Chlorination	
Pathogen	**Min.**	**Most Likely**	**Max.**	**Min.**	**Max.**	**Min.**	**Most Likely**	**Max.**	**Most Likely**	**Min.**	**Max.**
Rotavirus	0.0	1.4	>6.0	0.5	2.0	>1.0		>3.0	>6.0	1.0	3.0
Cryptosporidium	0.2	2.8	>6.0	0.5	1.0		>3.0		>6.0	0.0	0.5
Campylobacter	>6.0	>6.0	>6.0	1.0	3.0	2.0		>4.0	>6.0	2.0	6.0

All values are $\log_{10}$ removal. Secondary treatment includes dual media filtration and coagulation. All treatment removals are from the *Australian Guidelines for Water Recycling* (NRMMC-EPHC, 2006).

Sampling the water produced from wells downstream of the recharge points and comparing analysis results to those of the stormwater before recharge made it possible to obtain data about removal of pathogens by passage through the aquifer. If the water recovered from the aquifer is intended for drinking, it may need further treatment for certain pathogens (Page et al., 2015b).

5.2 METALS IN STORMWATER

The concentration of metals in urban stormwater runoff depends on the surfaces and materials over which the stormwater flows, along with time of year; traffic volumes; other sources of metals; and rainfall volumes, frequency, and event characteristics (Masoner et al., 2019; Pitt et al., 1994; Song et al., 2019; Weiss et al., 2008). Land use and climate also affect metals concentrations in runoff, and the NSQD allows for data filtering by land use and EPA rainfall zones (Pitt et al., 2015).

Roads are a common source of metals contamination due to vehicle activity that can deposit heavy metals. Metals can originate from components of motor vehicle exhaust, fuel leaks, detritus from tire and brake wear, detritus from road surface degradation, debris, and detritus from maintenance (Song et al., 2019). Table 5-2 summarizes data on selected metals in stormwater sampled from cities across the United States.

Table 5-2. Summary of Selected Metals in Stormwater Runoff (Adapted from Song et al., 2019)

Location	Zn (mg/L)	Pb (mg/L)	Cu (mg/L)	Cd (mg/L)	Fe (mg/L)	Mn (mg/L)
Texas	0.15	0.011	—	0.024	1.5	—
California	—	0.017	—	0.0094	—	—
Ohio	0.46	0.037	0.043	0.005	4.2	0.32
Maryland	1.2	0.22	0.11	0.035	—	—
Los Angeles, CA	0.51	0.033	0.93	0.0025	—	—

Infiltration of stormwater through practices such as EAR has been shown to be effective in controlling problems with urban runoff quantity and quality (Pitt et al., 1994). Natural filtering and adsorption are the two primary processes by which infiltration improves water quality (Song et al., 2019). However, as in many other filtration settings, contamination can still occur (Song et al., 2019).

Common heavy metals found in stormwater include lead, zinc, copper, and cadmium. A primary source of lead contamination in stormwater is old weathered paints (Bannerman et al., 1996; U.S. EPA, 1983). As stormwater flows across surfaces covered with weathered paint, which include buildings and roadways, it picks up some of the lead. The lead can be present in concentrations that exceed the 15 μg/L drinking water action level for lead. The primary lead removal mechanisms for stormwater are sorption, ion exchange, and precipitation (Pitt et al., 1994). Three studies were performed to assess lead removal by percolation of stormwater through rain garden column reactors in a laboratory (Davis et al., 2001; Hsieh and Davis, 2005; Sun and Davis, 2007; Weiss et al., 2008). These experiments found lead removal ranging from 62% to more than 99%. Removal was largely dependent on sand content—less lead was removed when sand content was higher. Field tests resulted in similar removal rates (Weiss et al., 2008).

Zinc is another common stormwater pollutant in parking lot, street, and roof runoff (NASEM, 2016; Pitt, 1996). The primary zinc stormwater removal mechanisms are precipitation, sorption, and ion exchange (Pitt, 1996; Weiss et al., 2008). Zinc has been found to be easily removed from stormwater during infiltration, having similar removal efficiencies to lead (Davis et al., 2001; Weiss et al., 2008).

Copper is commonly found in stormwater (Pitt et al., 1995; Weiss et al., 2008). Street and highway runoff appear to be primary sources (Pitt et al., 1995; Weiss et al., 2008). The primary mechanisms for removing copper from stormwater are sorption, complex ion formation, and ion exchange (Pitt et al., 1995; Weiss et al., 2008). While laboratory studies found copper removal efficiencies to be similar to those for lead and zinc, field studies found removal to be substantially lower (Davis et al., 2003; Weiss et al., 2008).

While cadmium is commonly detected in stormwater, it is most commonly detected at very low concentrations (Pitt, 1996; Weiss et al., 2008). Davis et al. (2001) suggested that wet deposition is likely the main source of cadmium in stormwater. One study found evidence to suggest that vehicle service runoff may be the most significant source of cadmium (Pitt et al., 1995; Weiss et al., 2008). The primary removal mechanisms for cadmium in stormwater are ion exchange, sorption, and precipitation (Pitt, 1996; Weiss et al., 2008). Laboratory studies have shown removal efficiencies in excess of 95% (Sun and Davis, 2007; Weiss et al., 2008), but due to relatively low concentrations in stormwater, field studies have not quantified cadmium removal (Weiss et al., 2008).

5.3 WATER QUALITY—ORGANIC COMPOUNDS

Organic compounds are a common stormwater contaminant that can affect groundwater quality in urban settings (Pitt et al., 1999). Organic contaminants are introduced to urban pavement such as roads and parking lots from a wide variety of sources and activities, including pesticide spraying and fluids dripping from motor vehicles, as noted above. Flow of stormwater over surfaces can contaminate stormwater before it reaches areas where it can infiltrate into groundwater.

5.3.1 Occurrence of Organic Contaminants in Stormwater

A recent, national scale-study provides a comprehensive characterization of organic contaminants in U.S. stormwater. Masoner et al. (2019) sampled 438 organic chemicals in urban stormwater from 50 storm events at 21 sites in 17 states. A total of 50 samples, one per storm event, were collected across the 21 sites. Some samples were filtered and others were not. The sites were in residential, commercial, and industrial areas with samples collected from stormwater infrastructure (concrete culverts and canals, or unlined dirt channels) that discharged mixed stormwater runoff from buildings, parking lots, roads, and other urban areas. Some limited data for organics can also be found in the NSQD (Pitt et al., 2015). However, the numbers of observations (a few hundred) are small compared to the number of storms in the database (>9,000 urban runoff events), and many results are non-detects.

Of the 438 organic chemicals analyzed, 215 (49%) were detected in one or more samples. The median number of organic chemicals detected per site was 73. Chemical concentrations (see Masoner et al., 2019, Table SI-9) were generally quite low; cumulative organic chemical concentrations of site samples ranged from 4,370 ng/L to 263,000 ng/L (median = 48,500 ng/L). Of the 215 organics detected, 69 (32%) were detected in more than 50% of samples. These 69 frequently detected organic chemicals accounted for 70% of all detections. Pesticides were the most frequently detected chemical group, with 35% of all detections, but accounted for only 5% of the total organic concentrations. PAHs accounted for 19% of total detections and 33% of total concentrations. Table 5-3, adapted from Figure 2 and Table SI-9 of the Masoner report, summarizes the top-occurring organics found by the study.

Table 5-3. Top Organics Identified in U.S. Stormwater Ranked by Detection Frequency and Concentration (Adapted from Masoner et al., 2019)

Name	Detection Frequency (%)	Name	Median Concentration (ng/L)
Pesticides			
Carbendazim	94%	Carbendazim	701
Fipronil desulfinyl	90%	Pentachlorophenol[1]	435
Diuron	86%	Diuron	51
Imidacloprid	86%	Piperonyl butoxide	43
PAHs			
Fluoranthene	90%	Benzo[b]fluoranthene	1,825
Pyrene	90%	Fluoranthene	1,590
Anthraquinone	88%	Chrysene	1,255
Phenanthrene	86%	Pyrene	1,250
Industrial Chemicals			
Methyl-1H-benzotriazole	92%	Bis-(2-ethylhexyl) phthalate[2]	2,235
p-Cresol	92%	Tri(2-butoxyethyl) phosphate	1,120
Tris(dichloroisopropyl) phosphate	82%	Methyl-1H-benzotriazole	861
Tri(2-butoxyethyl) phosphate	80%	4-Nitrophenol	796

[1] The Maximum Contaminant Level (MCL) for pentachlorophenol is 0.001 mg/L, or 1,000 ng/L. Unless noted, contaminants listed in this table do not have MCLs.

[2] The MCL for bis-(2-ethylhexyl) phthalate is 0.006 mg/L, or 6 ng/L. Unless noted, contaminants listed in this table do not have MCLs.

Pitt et al. (1995) collected and analyzed 87 stormwater source samples across a range of local source areas in Birmingham, Alabama. Ten pesticides and 16 PAHs were analyzed; one pesticide and 11 PAHs were found in more than 10% of the samples. The most commonly detected organics were 1,3-dichlorobenzene, fluoranthene, and pyrene (found in 20%, 20%, and 17% of non-filtered samples, respectively); seven other PAHs were found in more than 10% of samples (e.g., Table 3 in Pitt et al., 1995). Mean concentrations of detections of the three top occurring organics at the seven sampling sites ranged between 0.5 to 130 µg/L. The only pesticide found in more than 10% of samples was, somewhat surprisingly, chlordane (11%), for which most uses in the United States had been banned in 1983. Many of the highest concentrations were from urban stormwater runoff samples collected at vehicle service and parking areas.

Pitt et al. (1999) also tabulate a qualitative list of organic chemical and compound abundance in stormwater based on the NURP study (U.S. EPA, 1983). The pesticides lindane and chlordane are noted with moderate abundance in stormwater (the other four pesticides were noted as low abundance). Other organics listed with high abundance include fluoranthene and pyrene, and with moderate abundance benzo(a)anthracene, bis-(2-ethylhexyl) phthalate, phenanthrene, and pentachlorophenol.

Borden et al. (2002) collected and analyzed 249 stormwater samples from 46 sampling locations across North Carolina to identify land use types potentially associated with higher detection frequency and concentrations of fuel oxygenates and aromatic hydrocarbons. A range of land use areas was sampled (open space, low- and medium/high-density residential, commercial, industrial, gas stations, mixed land use). Seven oxygenates (including MTBE) and the benzene, toluene, ethylbenzene, and xylene (BTEX) toxicants were analyzed. Open space and low-density residential land uses had the lowest detection frequency and the lowest maximum concentration for most contaminants. All locations with significantly higher concentrations were associated with runoff from gas stations (including gas stations with leaky USTs). The highest oxygenate median concentration was for MTBE (1.29 µg/L) and the highest BTEX median concentration was for toluene (0.15 µg/L).

Whittemore (2012) investigated stormwater in six representative urban sand pits in residential areas in Wichita, Kansas. The broad suite of compounds sampled included 118 pesticide and pesticide degradate compounds, as well as 134 other organic compounds (plus other inorganic and bacteriological parameters). Nineteen pesticide or pesticide degradates were detected in the pit waters, including four (atrazine and its degradate deethylatrazine, metolachlor, alachlor, and acetochlor) of the five agricultural herbicides most often detected in U.S. stream waters. Also detected were all five of the herbicides (simazine, prometon, tebuthiuron, 2,4-D, and diuron) widely used for nonagricultural purposes in suburban and urban areas that are most commonly detected in U.S. stream waters. Concentrations of all pesticides detected were at levels significantly below EPA Maximum Contaminant Levels (MCLs); only one pesticide (atrazine) was measured at a concentration that exceeded one-tenth of its MCL (3 µg/L). Six non-pesticide organics were found in the pit stormwater (and 19 in groundwater at the site). The paper does not identify the six found only in the pit water, but organics detected in pit and groundwater include various phenols, fluoranthene, pyrene, and various volatile organic compounds (benzene, MTBE, and chlorinated ethane, propane, ethene, and benzene). All organics detected were found at very low concentrations substantially below drinking water standards, goals, or health advisories.

While not included in earlier studies of stormwater quality, the group of compounds known as per- and polyfluoroalkyl substances (PFAS) are increasingly being monitored in urban stormwater, in part due to the challenges of treating PFAS compounds in drinking water. In a study assessing green infrastructure technologies applied to urban stormwater, U.S. EPA (2018b) provide an extensive list of potential organic contaminants, including PFAS compounds, in urban stormwater in that study's Appendix Table A1. Xiao (2012) found PFAS detections in all seven urban stormwater samples in a study in the Minneapolis/St. Paul metropolitan area. They concluded that perfluoroalkyl acids (PFAAs) in stormwater runoff from residential areas derive mainly from rainfall, and that non-atmospheric sources at both industrial and commercial areas also contributed PFAAs in stormwater runoff.

In a study assessing PFAS in stormwater in Australia, Page et al. (2019) found that total concentrations of PFAS in stormwater runoff differ from event to event and were found to range from 14.3 to 96.0 ng/L. Perfluorooctane sulfonic acid (PFOS) and perfluorooctanoic acid (PFOA) were the most abundant PFAS in stormwater runoff. Trace organics are now being identified with increasing frequency in groundwater mostly due to improving analytical technologies that provide the ability to detect trace organics in extremely low concentrations (Maliva, 2020). Subsurface attenuation of PFAS is an active area of research. PFAS can be neutral, anionic, cationic, and even zwitterionic, and therefore interact with the soil matrix in a difficult to generalize fashion (Mejia-Avendaño et al., 2017). Several studies over the past decade focus on attenuation from sorption reactions in the subsurface; however, due to PFAS's complex properties (e.g., hydrophobicity or hydrophilicity), the solid–air interface and interactions with subsurface non-aqueous phase liquids (if present) can also play major roles in their fate and transport (Brusseau, 2018).

5.3.2 Fate and Transport of Pesticides

Pesticides are used heavily in urban areas to control weeds and insects in and around houses and along roads, railroads, parks, and private lawns. This has resulted in widespread pesticide contamination of U.S. groundwater. Detection, mobility, and removal of this pesticide contamination has been closely studied and characterized in Florida, California, and Arizona. Once pesticides are in groundwater, their mobility is highly variable, depending on the properties of individual pesticides and on soil conditions. Pesticides are found to be especially mobile through soils that are lacking clays. Pesticides with low water solubilities are less mobile. Pesticides also decompose in soil and water, but the amount of time needed for total decomposition can range from days to years (Pitt et al., 1999).

5.3.3 Fate and Transport of Other Organic Compounds

Other organic compounds are often found in groundwater in trace concentrations near urban stormwater recharge basins or wells. Concentrations of these organic compounds are often significantly reduced by percolation through soil. As with other pollutants, contamination of groundwater by organic compounds occurs more often in areas with transmissive sediments such as sand and gravel and in areas with shallow groundwater (Pitt et al., 1999).

Many organic compounds also readily volatilize into the atmosphere, including from groundwater (Pitt et al., 1999). Rates of volatilization are affected by a compound's concentration in groundwater, its physical and chemical properties, and site-specific soil and geological conditions (Pitt et al., 1999). Sorption is another significant attenuation process, affected by chemical and physical characteristics of both the sorbate and the sorbent (Pitt et al., 1999). Degradation and decomposition are also significant removal processes (Pitt et al., 1999).

5.4 WATER QUALITY—OTHER

5.4.1 Nutrients

In urban and suburban settings, nonpoint sources of nitrogen and phosphorus can include construction sites, lawn fertilizers, pet wastes, wildlife, leaf litter, grass clippings, and inputs from unsewered developments (Yang and Lusk, 2018). The contribution of nitrogen from atmospheric deposition is variable but can be significant (NASEM, 2016; Yang and Lusk, 2018). Roadway runoff has also been cited as a major source (e.g., Pitt et al., 1999).

Nutrients are routinely detected in stormwater runoff (e.g., Makepeace et al., 1995; Masoner et al., 2019; Pitt et al., 1999; Yang and Lusk, 2018). Work analyzing the nutrient content in stormwater has been underway for decades. The first comprehensive effort was NURP, which monitored stormwater discharges for a variety of pollutants in 28 cities across the United States from 1979 to 1983. Currently, the NSQD provides data on nutrients (Pitt et al., 2015). A simple comparison of overall nutrient concentrations from NURP and NSQD tabulated by Pitt and Maestre (2005) suggested similar values between the two datasets.

More recently, an extensive 2018 literature review and data compilation by Yang and Lusk (2018) includes a tabulation of information from 18 studies on the concentrations/loads and potential sources of nutrients in urban waters in the United States. The nutrient forms measured in stormwater characterization studies include ammonia, nitrite+nitrate, total nitrogen, total phosphorus, total dissolved phosphorus, and total Kjeldahl nitrogen. Results from the studies reviewed by Yang and Lusk (2018) illustrate the range of values among the studies. For example, mean total nitrogen concentrations range from less than 1 mg/L to more than 10 mg/L; mean nitrite+nitrate or nitrate values ranged from 0.01 mg/L to 7.0 mg/L, with many under 1 mg/L. Mean total phosphorus values were generally under 3 mg/L, with two studies reporting much higher values (means of up to 363 mg/L).

The contribution of nitrogen from atmospheric deposition is variable but can be significant (NASEM, 2016; Yang and Lusk, 2018). Other urban sources widely acknowledged include chemical fertilizers, pet waste, leaf litter, and grass clippings (Yang and Lusk, 2018). Roadway runoff has also been cited as a major source (e.g., Pitt et al., 1999).

5.4.2 Road Salt

Road deicing agents have emerged as a prominent stormwater concern in cold climates, as their use has been attributed to high chloride levels in some shallow aquifers in urban areas (e.g., Kelly, 2008;

Marsalek, 2003). Mixtures containing sodium chloride or calcium chloride are used the most, but alternative deicers such as calcium magnesium acetate and glycols are also used (Novotny et al., 1999). Road salt mixtures may also contain anti-caking additives (e.g., ferric and sodium ferrocyanide) and corrosion inhibitors (e.g., chromate and phosphate-based products) (Clark et al., 2010; Novotny et al., 1999).

Concentrations of constituents from road salt, primarily chloride and sodium, will vary according to usage, stormwater management, storm intensity, and other factors. They can, however, be quite high, and may continue beyond the snow season into the spring when residues are washed off the roads. Makepeace et al. (1995) note that stormwater chloride concentrations can range from 0.30 mg/L in snow to 25,000 mg/L when road salts are carried in runoff. Chloride concentrations in urban snowmelt in Syracuse, New York, have been measured as high as 17,200 mg/L (Novotny et al., 1999). A 2006 Water Environment Research Foundation project (Clark et al., 2010) found that sodium chloride concentrations can reach 10,000 mg/L in snow and ice from urban streets. A case study in Maliva (2020) reports that chloride concentrations of up to 1,100 mg/L have been found in parking lot runoff in Suffolk and Nassau Counties in New York.

5.4.3 Trace Organics

The term "trace organics" refers to various emerging organic contaminants for which the environmental and health effects are not well established. This includes contaminants such as pharmaceuticals, antibiotics, synthetic and natural hormones, personal care products, detergent metabolites, antimicrobial agents, brominated flame retardants, perfluorooctane surfactants, fragrance and flavoring compounds, insect repellants, x-ray contrast agents, plasticizers, and caffeine (Maliva, 2020). Many of these contaminants are widely dispersed, introduced to the environment anywhere humans go—including urban surfaces and soils that generate stormwater runoff (Maliva, 2020).

The effects of trace organics in the environment are an active area of research. In laboratory studies, Alidina et al. (2014a) evaluated whether pre-exposure of the soil microbial community to trace organics affects microbial attenuation of trace organics (and found no indication that it does). Related studies concluded that substrate composition has a larger effect than trace organic concentration on attenuation (Alidina et al., 2014b). Another study focused on the effects of temperature on the attenuation of trace organics (Alidina et al., 2015), finding that:

- Only six of the 22 trace organics evaluated showed changes in attenuation dependent on temperature.
- Attenuation of four trace organic compounds (diclofenac, gemfibrozil, ketoprofen, and naproxen) decreased as temperature dropped, likely due to decreased microbial activity.
- Attenuation of oxybenzone and trimethoprim increased at lower temperatures (Alidina et al., 2015).

5.5 MOBILIZATION OF SUBSURFACE CONTAMINANTS

While many studies show that most contaminants are removed from infiltrating stormwater in the upper soil layers (Dallman and Spongberg, 2012; Pitt, 1996; Weiss et al., 2008), water quality differences between infiltrated stormwater and ambient water in an aquifer may also result in mobilization of subsurface contaminants (Dallman and Spongberg, 2012; Maliva, 2020; Song et al., 2019; Vanderzalm et al., 2016). Infiltrating stormwater can result in the following reactions and processes in the subsurface (Maliva, 2020; Song et al., 2019):

- Mixing of recharged and native groundwater
- Precipitation or dissolution of minerals
- Oxidation reactions caused by introduction of dissolved oxygen into chemically reducing aquifers
- Reduction reactions caused by microbial activity if the stormwater contains nutrients or high organic carbon
- Sorption/desorption and cation exchange reactions
- Clay swelling and dispersion

As recharge water and ambient waters mix, mineral dissolution or alteration, desorption from aquifer solids, and cation exchange reactions can release ions, including arsenic and other metals, into groundwater, adversely affecting water quality—although physical attenuation due to dilution and dispersion can also improve the quality of recharge water (Maliva, 2020). Decreased pH will promote the desorption and mobilization of heavy metals from sediment iron (III) minerals in an oxidized environment, although it will promote the adsorption of arsenate (Fakhreddine et al., 2020). Decreased Ca^{2+} and Mg^{2+} concentrations will promote desorption of arsenate from clay minerals (Fakhreddine et al., 2015). The introduction of oxidized water into an anoxic aquifer can promote the release of arsenic through the oxidation of the minerals pyrite and arsenopyrite (if present in the sediment), although arsenic mobilization can also be limited by adsorption onto clays and iron (hydr)oxide minerals (Fakhreddine et al., 2015; Neil et al., 2014). See Section 6.3.2 for further discussion of these issues.

The swelling and dispersion of clay minerals due to decreased cation concentrations in the water (especially Ca^{2+} and Mg^{2+}) can clog affected strata, dramatically reducing hydraulic conductivity (Maliva, 2020). In ASR applications, recharge flow rates and recovery flow rates are affected by such changes in hydraulic conductivity when they occur (Maliva, 2020). Given the diversity of potential effects on aquifer properties, the geochemical processes influenced by recharging stormwater are highly site-specific (Maliva, 2020). See Section 6.3.2 for further discussion of EAR best practices affected by site geochemistry.

The availability of water quality data is affected by which parameters are specifically required by regulatory agencies in permits because permit requirements often focus on parameters for which drinking water standards have been established (Maliva, 2020). Some important water quality parameters, such as calcium, magnesium, and bicarbonate, are often not sampled, though their presence is reflected in measurements of total dissolved solids. Table 5-4 lists a set of parameters useful for a thorough geochemical compatibility analysis.

Geochemical processes at EAR sites can often be better understood when sampling is supplemented by geochemical modeling. Widely available software (e.g., the USGS program PHREEQC; Parkhurst and Appelo, 2013) allows use of recharge and ambient groundwater quality data, along with information on the mineralogy of the recharge zone, to conduct simulations such as 1) predicting the composition of the mixed, equilibrated water (Mirecki et al., 2013); 2) determining the saturation state of the ambient groundwater, recharge water, and mixed water with respect to sediment mineralogy; and 3) predicting changes in the geochemistry over time, including mineral dissolution and precipitation. Microbially mediated processes can be incorporated into geochemical modeling. Also, many geochemical models allow some limited transport modeling (1-D or 2-D) to explore these geochemical changes along a groundwater flow path.

Coupling geochemical modeling with 3-dimensional fluid flow simulation, reactive transport models can be used to analyze EAR scenarios. For example, to model processes in the variably saturated conditions of the vadose zone, the program Hydrus-1D has been coupled with PHREEQC to form HP1 (Jacques and Simunek, 2005). The USGS program PHAST, another example, simulates transport in the saturated zone. It combines a version of PHREEQC with the flow and transport program HST3D (Parkhurst et al., 2010).

Table 5-4. Useful Parameters for Geochemical Analysis of Water

Parameters	Relevance to EAR
Sodium Chloride Sulfate Total dissolved solids	• These parameters relate to salinity. • Depending on the aquifer, and especially if the aquifer is used for drinking water, there may be concerns about recharging high-salinity stormwater.
Calcium Magnesium Bicarbonate (alkalinity) pH	• These parameters relate to clogging potential and water quality. • Sources of recharge water with high calcium, magnesium, and carbonate concentrations or high pH may cause precipitates to clog the EAR system. • Sources of recharge water with low pH may cause mobilization of metals from the soil or sediments into groundwater.
Oxidation reduction potential/E_h Dissolved oxygen Dissolved iron (Fe^{2+}) Dissolved manganese (Mn^{2+})	• These parameters indicate the redox status of the system, an important control on water chemistry. • High redox potential and high dissolved oxygen concentration accompany low to no dissolved iron and manganese, as these metals will remain in mineral form. • Low redox potential and dissolved-oxygen-free conditions accompany the release of iron and manganese from the oxide/hydroxide minerals into groundwater.
Arsenic Uranium Molybdenum Nickel Zinc Cobalt	• These parameters relate to water quality and are generally considered contaminants in groundwater. • Some of them (arsenic, uranium, molybdenum) are sensitive to redox status. • Nickel, zinc, and cobalt can desorb from sediments due to low pH or be released due to dissolution of iron and manganese minerals under low E_h.

Parameters	Relevance to EAR
	• Pyrite and arsenopyrite in the sediments oxidize under high E_h conditions, releasing arsenic.

6 BEST PRACTICES

Stormwater EAR is a potentially effective and safe way to augment water supplies, but also can present a risk of groundwater contamination. While stormwater EAR is still an active area of research, this section discusses what current scientific understanding suggests about best practices to advance effective and safe stormwater EAR under diverse development and hydrogeologic conditions.

6.1 SITE SELECTION

The overall success of an aquifer recharge project hinges on the selection of an appropriate site for a project's purpose. Stormwater EAR systems can be used for multiple purposes including restoring depleted aquifers, enhancing drinking water supplies, providing seasonal storage for later uses such as irrigation or drinking water supply, mitigating saltwater intrusion impacts, or reducing flood risks. Each of these goals has unique requirements for, and constraints on project design, operation, and maintenance that depend directly on-site conditions. The following factors affect a potential site's viability for stormwater recharge:

- Areal extent, thickness, and depth of the aquifer (available storage space)
- Site geology, geologic structures, and degree of homogeneity/heterogeneity (including presence of fractures, joints, solution conduits)
- Aquifer properties (storage potential, transmissivity, storativity, porosity, hydraulic conductivity, unsaturated thickness)
- Thickness of potential confining layers
- Aquifer geochemical properties (mineralogy)
- Chemistry of the native groundwater
- Chemistry of stormwater
- Hydraulic gradient, groundwater velocity
- Wet season that occurs during periods of lower demand
- Proximity of the recharge point to other wells or boreholes
- Proximity to potential and actual sources of contamination and contaminant plumes
- Proximity to stormwater collection system
- Proximity to an entity in need of recovered water (for ASR operations)
- Availability of stormwater (or a combination of stormwater and other sources of water) to be recharged

Aquifer recharge projects are implemented in the United States across a range of site settings and for a range of purposes. Although Section 4 discusses several examples of stormwater EAR, projects that recharge stormwater are still relatively uncommon in comparison to AR and ASR projects that recharge reclaimed wastewater, groundwater, or surface water (Bloetscher, 2015; Shaw et al., 2020; Stefan and Ansems, 2018). In an inventory of roughly 1200 EAR projects, only 6% of the 288 identified projects in the United States use stormwater as their influent source (Stefan and Ansems, 2018). In a working database of ASR projects maintained by the American Water Works Association, stormwater as a source is grouped with all surface waters (Bloetscher, 2015). Still, lessons can be learned from surface water recharge systems given overlap in surface water quality and stormwater quality and commonalities in site selection drivers.

Table 6-1 presents selected sample projects across the United States where ASR operations using surface water have been tested or operated (Bloetscher, 2015; Brown et al., 2006; Shaw et al., 2020). Projects are implemented in various geologic settings, for various purposes, and experience a range of problems from physical clogging to water quality concerns. Brown et al. (2006), who collated observations from 50 projects including some of those included in Table 6-1, summarize a number of lessons learned from their study. First, they found that clogging was a common concern in all geologies, though it was becoming less severe as designers and operators learn from past projects. They note that helpful remediation efforts for clogging include regular monitoring of specific capacity or injectivity and routine backflushing, which generally has to be performed more often in sand aquifers than in karstic ones. They also found water quality concerns owing to geochemical reactions to be rather common. Lastly, Brown et al. (2006) stress the importance of installing adequate monitoring equipment, including redundancies, to ensure operators can stay ahead of any clogging or water quality concerns. These best practices and others, derived from the stormwater EAR and broader recharge literature, are discussed throughout Section 6.

Table 6-1. Selected Aquifer Storage and Recovery Operations in the United States (Adapted from (Bloetscher, 2015; Brown et al., 2006; Malcolm Pirnie, Inc. et al., 2011; Shaw et al., 2020)

Location	Primary Purpose	Aquifer Type	Recovered Water Quality Issues
Calleguas, CA	Emergency water supply	Sand	Minor—low concentration of manganese, iron
Highlands Ranch, CO	Meet seasonal demands	Sandstone	None
South Denver, CO	Meet seasonal demands	Sandstone	Minor—biological growth in wells
Washoe County, NV	Meet seasonal demands	Glacial sand	None
Beaverton, OR	Meet seasonal demands	Basalt	None
Portland, OR	Meet seasonal demands	Sand and gravel	None
Salem, OR	Emergency water supply	Basalt	Minor—disinfection byproducts and natural radon
Hilton Head, SC	Meet peak and seasonal demands	Limestone	None
Myrtle Beach, SC	Meet seasonal demands	Sand	Minor—low concentration of manganese, iron
Huron, SD	Restore aquifer levels	Glacial sand	Minor—atrazine in recharge water
Kerrville, TX	Supplement surface supplies	Sand	None
Salt Lake City, UT	Meet seasonal demands	Sand and gravel	None
Seattle, WA	Meet seasonal demands	Sand and gravel	Minor—radon
Green Bay, WI	Meet peak demands	Sandstone and limestone	Major—arsenic, manganese, and cobalt
Oak Creek, WI	Meet peak and seasonal demands	Sandstone	Minor—low concentration of manganese, iron

6.1.1 Suitability Mapping

Given the multitude of spatially variable factors that dictate site suitability, geographic information systems (GIS)–based multi-criteria decision analysis (MCDA) studies are often used to develop suitability maps for EAR systems. Sallwey et al. (2019) reviewed 63 studies that applied GIS-MCDA for EAR site selection. They found slope to be the most commonly included criterion, and geology and hydrologic soils to be the highest weighted, but they found no common approach to suitability map generation. Nevertheless, general trends can be seen with a narrower scope and a focus on suitability mapping efforts in the context of region-specific goals. For example, the Texas Water Development Board conducted a survey of the state's aquifers to determine their relative suitability for use in aquifer recharge or ASR projects (Shaw et al., 2020). They developed a GIS-based screening procedure and a final suitability rating based on three categories of criteria: hydrogeological parameters, excess water availability, and water supply needs. The resulting suitability map assigns each grid cell (90 miles by 90 miles) a value of 0 to 1, which managers can use to categorize areas and aquifers with high, medium, or low suitability. The authors conclude that the suitability map is an important screening-level tool that stakeholders can use as an indicator of the probability of finding a suitable site for recharge, including ASR. They are also clear about the map's primary limitation, in that recharge and ASR projects are inherently site-specific and the map is not a substitute for a thorough, site-specific feasibility evaluation (Shaw et al., 2020).

Similar suitability mapping efforts have been conducted in California's Central Coast, with regional variations that captured benefits uniquely important to the local communities. Russo et al. (2015) developed a regional suitability map for the Pajaro Valley Groundwater Basin based on surface and subsurface spatial datasets, including land cover, soils, and geologic characteristics. They combined the suitability map with a regional groundwater model to assess the effectiveness of hypothetical infiltration practices in restoring groundwater levels and mitigating saltwater intrusion. Although they too concluded that their results were best used as a relative guide and were not a substitute for site-specific evaluations, their approach provided reasonable guidance for the siting of future recharge practices and estimated regionally important benefits that could inform future management decisions. Fisher et al. (2017) developed a similar suitability map for Santa Cruz and northern Monterey Counties based on surface and subsurface spatial datasets, but instead combined the map with a runoff response model. This variation allowed the authors to identify locations where stormwater runoff from hillslopes could be used for distributed stormwater capture systems with benefits on the order of 100–1,000 acre-feet/year of recharge, an intermediate scale between low-impact development and highly engineered regional MAR systems. This variation, which incorporated an explicit water supply component, was important to the local study communities as they rely on local groundwater supplies for roughly 85% of their freshwater needs (Fisher et al., 2017).

6.2 SURFACE SOIL CHARACTERIZATION

For passive infiltration practices like infiltration basins, characterization of soil hydraulic conductivity is necessary to ensure effectiveness. Designers can use infiltrometers to estimate infiltration rates of in situ native soil and provide a measure of soil hydraulic conductivity. Soil hydraulic conductivity is often highly heterogeneous both horizontally and vertically (Racz et al., 2012). Because of this, designers must take care during preliminary site characterization to collect a sufficient number of samples and interpret findings appropriately.

Variable conductivities do not, however, always lend themselves to variable infiltration rates. For example, in the characterization of sites for dry wells in eastern Washington, Massmann (2004) notes that estimated hydraulic conductivity values—which only measure small volumes of soil—varied by approximately three orders of magnitude, while observed infiltration rates only varied from 0.2 to 2 cfs. Planners should also note that infiltration rates may be subject to large seasonal and temperature variations (Constantz et al., 1994; Emerson and Traver, 2008; Jaynes, 1990; Ronan et al., 1998; Schuh, 1990) and should accommodate these fluctuations in their designs accordingly.

6.3 AQUIFER EXTENT AND SITE GEOLOGY

The suitability of a site for EAR depends on many factors including the size and storage capacity of the aquifer, presence of a confining layer, site access, and a range of other hydrogeologic parameters. These factors also have varying influence depending on the purposes of the project. A site with an unconfined aquifer may in some cases be more desirable than one with a thick layer of impermeable material, such as clay, over a confined aquifer. While the unconfined aquifer could be recharged using shallow spreading basins or distributed infiltration practices (e.g., the distributed infiltration practices in Los Angeles and Nassau County described in Table 4-2), the confined aquifer might only be able to be recharged using injection wells, which are generally more expensive to construct than other passive infiltration systems (Maliva, 2020)—although the cost of acquiring land for a spreading basin may be relevant. Still, if the project is intended to store large volumes of water for subsequent recovery, confined aquifers can be suitable candidates (e.g., the Parafield site described in Table 4-2). The material constituting the aquifer also effects desirability. Intergranular dominated aquifers, such as those composed of coarse sands or gravels, are generally the best unconsolidated aquifer materials for EAR, while sandstones, conglomerates, and limestones are generally the most preferable consolidated aquifers for these projects. These aquifer types generally have ample effective porosity and hydraulic conductivity, indicators of the aquifer's available storage and recharge rate.

6.3.1 Aquifer Hydraulic Properties

Aquifers need to be well-characterized to be considered for EAR operations. The important properties depend on the project purpose; among them can be the aquifer's storage capacity, transmissivity, conductivity, and accessibility, each of which can be difficult to characterize. Aquifer recharge projects, especially those with recovery in mind, might call for a greater degree of aquifer characterization than typical hydrogeologic assessments (Behroozmand et al., 2019; Maliva et al., 2015). For ASR projects, recovery of injected water is also important. Predicting recharge project performance often requires the use of robust groundwater flow models that incorporate aquifer heterogeneity (Maliva et al., 2015). For projects that recharge stormwater into freshwater aquifers, standard hydrogeologic techniques may be appropriate for aquifer characterization, but in systems in which freshwater is injected into brackish or saline aquifers, greater characterization of the movement of the injected water is needed (Maliva et al., 2015).

Standard borehole logging techniques, such as caliper, natural gamma ray, spontaneous potential, electrical resistivity, sonic, fluid conductivity, temperature, and flow meter logging, can provide coarse-scale data on aquifer heterogeneity (Maliva, 2020). But advanced borehole logging techniques, such as nuclear magnetic resonance (NMR), microresistivity imaging, and gamma ray spectroscopy, can be used in EAR projects to provide finer-scale porosity and pore-size data to assess the potential feasibility of the site (Maliva et al., 2015). Surface hydrogeophysical methods, such as resistivity and electromagnetic methods, ground-penetrating radar (GPR), surface NMR, seismic reflection and refraction, and relative gravity surveys may also be used to characterize aquifers, but they provide less vertical resolution than borehole logging techniques (Maliva, 2020). Other researchers have developed a towed time-domain electromagnetic system to create lithography maps and assess the suitability for EAR (Behroozmand et al., 2019).

The storage capacity of aquifer material is generally expressed using parameters such as storativity, specific storage, and specific yield, depending on the aquifer type and the analysis conducted (Maliva, 2020). Similarly, aquifer pore space, which affects the percentage of the aquifer potentially available for water storage, can be characterized by measurements of total porosity, effective porosity, and specific yield. The different terms capture the differences between a physical property (e.g., total porosity) and how that physical property influences interactions with water (e.g., effective porosity and specific yield). For example, the total porosity, effective porosity, and specific yield of coarse-grained, granular rock are all close to each other. In comparison, clays tend to have relatively high total porosity but with substantial pore space that is discontinuous. This impedes flow, rendering some pore spaces inaccessible and resulting in a lower effective storage capacity and specific yield (Maliva, 2016, 2020). Lastly, storage capacity depends on the amount of pore space not occupied by water. The greater the unsaturated thickness above an unconfined aquifer, the more storage is available. Areas of the United States with high water demand often have aquifers with substantially more unsaturated thickness during high demand periods.

Hydraulic conductivity is a measure of how freely water flows through aquifer material. Stormwater is often produced in large volumes over relatively short periods, requiring efficient means of directing it to storage locations. Aquifers with higher conductivity are more desirable for EAR for flood risk reduction because water can be recharged quickly. For ASR applications, however, very high hydraulic conductivities may move recharged water quickly away from the recharge location, reducing recovery efficiencies.

For ASR applications, a relatively homogeneous aquifer is conducive to recovering a large percentage of the volume of water recharged (Maliva et al., 2015). In contrast, highly heterogeneous aquifers lead to greater dispersive mixing between the recharged water and the native groundwater—a potential problem in saline or brackish aquifers (Maliva, 2020). In karst and fractured aquifers, recharged water quickly travels away from ASR wells, causing mixing with the native groundwater, and reducing the recovery efficiency (Maliva, 2020; Page et al., 2011). Similarly, fractured volcanic rock aquifers are generally not suitable for ASR applications (Wolcott, 1999). On the other hand, relatively homogeneous sand aquifers are likely to provide high recharge efficiency for ASR systems. Also, if a storage zone is not well confined, recharged water may flow out of the storage zone, and saline water may flow into the storage zone during recovery (Reese, 2002).

6.3.2 Geochemistry

There is an interplay of physical, geochemical, and biogeochemical processes when stormwater is introduced into an aquifer. The effects of interactions between geochemical properties of the aquifer, the quality of the ambient groundwater, and the chemistry of recharged water on the performance of EAR projects are discussed in Sections 5.2 and 5.4. Good site characterization, an understanding of the significant processes, modeling if appropriate, and well-planned monitoring are needed to anticipate and mitigate problems with EAR projects. In some cases, stormwater may contain contaminants or other water quality characteristics that require pretreatment.

Characterization of the native groundwater quality, stormwater quality, the soil and sediment in the vadose zone, and the sediment in the saturated zone are needed to assess the potential for 1) contaminant removal from the infiltrating stormwater, 2) the expected fate of contaminants in the aquifer, 3) the potential for mobilizing metals from aquifer sediments, and 4) the potential for clogging or changes in hydraulic properties due to geochemical and biogeochemical processes.

Heavy metals (e.g., Cu, Cd, Cr, Pb, and Zn) may be removed in significant quantities during infiltration through the soil (e.g., Yousef et al., 1990). Metals associated with small particles may be physically retained in the soil during infiltration. Metals dissolved in the water may be retained in the soil via ion exchange, sorption, or precipitation in secondary minerals, although some dissolved metals can reach the aquifer. When stormwater is injected directly into aquifers, there is no soil column available to provide contaminant removal before water reaches the saturated zone. Factors affecting metals removal include clay content, organic matter content, the presence of iron and manganese minerals, the speciation (forms) of the metals, the amount associated with particles, and pH of the water. Unlike organic contaminants, metals do not biodegrade. Therefore, pretreatment (e.g., to remove metals, adjust pH, provide other chemical pretreatment, or remove particulates; see Section 6.7) or retention in the soils or aquifer sediments are needed to limit metals reaching and migrating in groundwater.

Organic contaminants vary widely with respect to properties such as biodegradability, solubility, and tendency to volatilize. Infiltration through soil can remove substantial quantities of organic compounds depending on contaminant and soil properties. Clay content is particularly significant for sorption of organic contaminants.

Consequences of differences between recharging stormwater and ambient water include the following redox-related effects:

- Arsenic has been noted as a significant water quality concern in some aquifer recharge systems (e.g., Neil et al., 2014). It can be released when the introduction of oxygenated water into an anoxic aquifer oxidizes the sulfide minerals arsenic and arsenopyrite. The mobilization of arsenic can be mitigated when iron released from oxidation of these sulfides oxidizes to form secondary iron oxide/oxyhydroxide minerals to which the ion arsenate can adsorb; the degree to which this happens will be site-specific and depend in part on water chemistry (Neil et al., 2014).
- Should E_h conditions in the aquifer drop to reducing conditions, existing iron and manganese oxide/hydroxide minerals will be dissolved, and any contaminants that were associated with these minerals (metals, organics, phosphorus) can be released.

Other water chemistry differences between the recharge and groundwater can include pH, ionic strength, and cation (Na^+, K^+, Ca^{2+}, Mg^{2+}) concentrations. A decrease in pH will promote the desorption of metals from sediment iron oxides/hydroxides but will promote the adsorption of arsenate (Fakhreddine et al., 2020). A decrease in the Ca^{2+} and Mg^{2+} concentrations in the water promotes desorption of arsenate from clays (Fakhreddine et al., 2015).

Physical changes may also occur. Decreased Ca^{2+} and Mg^{2+} in the water causes clays (smectites) to swell and disperse. This swelling and dispersal can cause clogging in the subsurface, reducing hydraulic conductivity. Adjustment of the recharge water quality to increase Ca^{2+} and Mg^{2+} concentrations could mitigate this effect. Conversely, if the mixing of the recharge water with groundwater changes the saturation status with respect to minerals in the aquifer (e.g., calcite), mineral dissolution could increase hydraulic conductivity. Mixing of oxic and anoxic waters can lead to clogging due to precipitation of iron (hydr)oxide precipitates (Medina et al., 2013). See Section 6.7.8 below for further discussion of chemical pretreatment options for addressing these issues.

The extent and rates at which processes discussed above occur, and other mitigating processes and factors will be site-specific. Also, geochemical heterogeneity in the aquifer (e.g., redox conditions) may cause variations in which processes occur in various parts of a recharge system (e.g., Vanderzalm et al., 2016; Warner et al., 2016). Stormwater chemistry can vary between storms as well as seasonally, making ongoing monitoring an important component of EAR system management, especially if pretreatment is warranted.

6.4 OPERATIONAL AND ECONOMIC CONSIDERATIONS

The economic drivers of EAR using stormwater include technical considerations such as the timing of precipitation and demand and the means to convey stormwater underground—factors include aquifer storage size, permeability, infiltration, injection and recovery rates, and connections with other aquifers.

> Based on analysis of a 70-acre site in New Mexico that drains to 11 acres of retention ponds, Miller (2006) suggests communities that implement rapid infiltration practices (about 1 foot/day minimum infiltration rate) can achieve significant savings in net water consumption.

EAR is often considered in areas where periods of high precipitation coincide with periods of lower water demand (Brown et al., 2006; Maliva, 2020). This creates a surplus of water, which may be available for aquifer storage, during wet periods, to be recovered during the high demand periods, when the available supply of surface water, for example, does not always meet demand (Brown et al., 2006). EAR using stormwater is also deemed more beneficial during wet periods because of its function in reducing flood risks.

Evaporation rates, land costs, and pumping costs also influence the economics of EAR. Arshad et al. (2014) conducted modeling to compare the factors that affect financial competitiveness when comparing EAR with surface water storage. They found that evaporative losses from surface storage of water can be on the order of 30–50%, which represents a considerable loss compared to EAR. However, the cost of injection and withdrawal pumping for subsurface storage was a relevant factor in determining which option was preferred at a site described by Arshad et al. (2014). Despite these additional pumping costs, the economic case for EAR can be compelling, especially when combined with important externalities. When evaluating options for supplementing quickly diminishing water supplies in Kerrville, Texas, a technical feasibility assessment found that using ASR wells to divert surface water into the local, depleted aquifer could yield capital expenditure savings of $26 to $30 million (in 1990 dollars) compared to constructing a traditional off-channel reservoir (Malcolm Pirnie, Inc. et al., 2011). Moreover, an ASR approach would eliminate the need to inundate and impact hundreds of acres of existing habitat. The Kerrville ASR system was constructed and currently supplies the town with around 10% of its water, especially during dry times (Malcolm Pirnie, Inc. et al., 2011).

High infiltration rates lower the overall cost of EAR. Uncertainty in standard parameters—infiltration rates, injection rates, etc.—must be considered as long-term fluctuations can have detrimental effects on cost, operational efficiency, and project effectiveness. The fact that subsurface storage in EAR reduces evaporative losses (as occur with surface storage of water) is a cost advantage for EAR. The need for infrastructure (whether a well or a maintained basin) under EAR can increase costs. Injection wells are typically much more expensive than spreading or infiltration basins (Bouwer, 1988; Bouwer et al., 2008; Maliva, 2020), although the cost and availability of land can affect the viability of spreading basins. Cost-benefit analyses for these decisions will be most robust when benefits can be monetized with reasonable assumptions and potential risks and uncertainties are incorporated (Maliva, 2014).

In an example of an economic analysis that highlights the potential value of EAR, Devinny et al. (2004) estimate a distributed system of infiltration basins in the Los Angeles region could increase infiltration rates by about 50% across the region, corresponding to a volume of about 300 acre-feet per year. Assuming a 90% recovery efficiency and comparing to the anticipated cost of desalination of $800 per acre-foot, the benefit of this recharge volume would be $216 million per year. Miller (2006) suggests state and local governments should provide return flow credits for households and communities that implement stormwater recharge practices, as this return flow directly offsets local abstractions. In some water-stressed areas that already pay a high price for imported water, the value of recharged groundwater may be greater. However, water rights issues may complicate the analysis when extraction of groundwater may be subject to legal challenges.

6.5 SALTWATER INTRUSION

Coastal aquifers are important natural resources, providing water supplies for many coastal communities and freshwater inputs to many coastal ecosystems (Burnett et al., 2003; Costall et al., 2020; Johannes, 1980; Moore, 2010). When overdrawn, these aquifers can be subject to saltwater intrusion, or the landward migration of the underground saltwater interface. Impacts from saltwater intrusion can range from salinization of supply wells to reduced freshwater discharges to coastal springs, rivers, and submarine habitats (Barlow and Reichard, 2010; Costall et al., 2020; Werner et al., 2013). Mitigation of saltwater intrusion impacts generally entails reducing withdrawal rates or increasing recharge rates. A wide body of literature exists on alternative prevention measures such as physical flow barriers, air injection barriers, injection wells, etc.: for example, see work by Luyun et al. (2011) and reviews by Essink (2001) and Werner et al. (2013), though these tend to be more complex and less reliable than measures that focus on restoration of the local water balance such as reduced withdrawals or EAR (Calvache and Pulido-Bosch, 1997; Essink, 2001; Werner et al., 2013).

Stormwater EAR can be an effective way to mitigate impacts from saltwater intrusion, though the measurement, modeling, or prediction of impacts and their mitigation is extremely difficult owing to myriad subsurface complexities (Costall et al., 2020; Werner et al., 2013). Monitoring wells are the traditional way in which saltwater intrusion is detected, though aquifer heterogeneity creates a large potential for observational error. Costall et al. (2020) combined numerical simulation, geophysics, and analysis of more than 30 years of data at one site to conclude that determining the landward extent of the seawater interface is extremely challenging.

Despite significant complexities, general conclusions can be drawn from studies that evaluate the effectiveness of EAR on saltwater intrusion mitigation. Russo et al. (2015) used a spatial evaluation of recharge suitability combined with a regional groundwater model to, in part, quantify the impact of infiltration-based EAR projects on saltwater intrusion. Their results suggest that EAR projects implemented near the coast can help reduce saltwater intrusion more rapidly, but that much of the recharged water will be lost to the ocean. In comparison, EAR projects placed further inland are slower to reduce saltwater intrusion but are more effective in the long term and result in less recharged water flowing to the ocean (i.e., more remains in coastal aquifers and available for other purposes). Another study simplified the process even further, suggesting that a long-term recharge to withdrawal ratio of 1.3 is necessary to keep seawater intrusion at bay (Misut and Voss, 2007).

Along the Atlantic Coast of Florida, historic withdrawals from the deep Floridan aquifer combined with an extensive network of surface drainage canals has led to widespread saltwater intrusion. To reverse these impacts, water resource managers have installed control structures in the canals and divert large volumes of inland surface water, supplemented with stormwater runoff, to these canals. This increases surface water stages along the coast which, combined with the highly permeable karstic geology of the area, helps create a barrier to further saltwater intrusion (Barlow and Reichard, 2010; Bouwer et al., 2008).

6.6 SOURCE WATER PROTECTION

To protect water quality in aquifers where urban stormwater is the major source of recharge water, a well-established concept, the source water protection area (SWPA), can be used to safeguard against water quality degradation caused by the introduction of contaminants from stormwater. Source water protection ordinances can be used to protect groundwater (and surface water) supplies by restricting land uses in a groundwater recharge area (or around a reservoir) used for drinking water. Source water protection has been used by drinking water utilities as one of multiple barriers to protect drinking water.

In general, SWPAs are delineated for surface water intakes (i.e., areas upstream of intakes) and groundwater wells (i.e., areas upgradient of wells) that contribute water (and thus contaminants) to these withdrawal points. Land uses and anthropogenic activities in the SWPAs can contribute undesirable contaminants to the stormwater. These contaminants must be appropriately treated before entering the aquifers (e.g., via infiltration basins and injection wells) to minimize risks. Although SWPAs for most drinking water utilities were delineated under the Source Water Assessment provision of the 1996 Safe Drinking Water Act Amendments, areas used to collect stormwater for subsequent recharge are not included under the 1996 Amendments. In other words, additional work would need to be done to delineate a stormwater capture area, and an inventory of land use and anthropogenic activities in the capture area would need to be developed to characterize risks associated with the use of stormwater to recharge aquifers.

In an urban setting, where significant areas are impervious, stormwater is generated rapidly and in high quantity. An approach to protect the quality of the stormwater will protect the quality of recharge water for stormwater EAR systems and thus the quality of water in the aquifers. Through an exercise similar to the Source Water Assessment program, entities can establish goals and objectives of their operations and use them to set quality criteria for recharge water. Depending on the nature, magnitude, and intensity of the activities in the stormwater capture areas, various management practices can be implemented to address recharge water quality issues. As noted in Section 6.7.3, the use of green and natural infrastructure can help with stormwater quality issues.

6.7 PRETREATMENT

Pretreatment of stormwater before infiltration or injection is often necessary for a variety of reasons. Depending on site conditions and project objectives, designers may incorporate pretreatment systems to reduce physical clogging of the recharge practice by sediment, reduce organics and nutrients to lessen the chance of biofouling, or reduce pollutants to ensure the quality of stormwater is suitable for recharge or recovery. Pretreatment systems are also typically designed to address the first flush of stormwater, which carries with it disproportionately high concentrations of contaminants.

Not all stormwater recharge systems use pretreatment, but if pretreatment is used, it may involve pre-sedimentation basins (Bouwer, 1988; Jeong et al., 2018; Maliva, 2020; Pedretti et al., 2012), constructed wetlands, green infrastructure (Abel et al., 2015; Bouwer, 1988; Hagg et al., 2018; Hartog and Stuyfzand, 2017; Jeong et al., 2018; Maliva, 2020; Page et al., 2014), media filtration (Bouwer et al., 2008; Lin et al., 2006; Pavelic et al., 2006), or coagulant or flocculant use. If a larger system such as a sedimentation basin is not feasible, even having stormwater enter the system over a concrete slab with rocks or coarse gravel around it for energy dissipation can be helpful (Bouwer et al., 2008; Maliva, 2020). In all cases, regular maintenance is critical to maintain expected function and water quality treatment performance. Operation and maintenance best practices are discussed further in Section 6.8.

Each pre-treatment approach is unique and the proper approach for a given practice will depend on a number of factors. Local stormwater codes may require a settling basin of a certain size be implemented prior to specific infiltration practices, local groundwater codes may require stormwater be treated to a certain quality prior to infiltration or injection, or local stakeholder preference may be for a pretreatment system like a constructed wetland to also provide an aesthetic amenity for the community. Ultimately, the design of a stormwater recharge system and its pretreatment will be site-specific. Below, we provide an overview of the major types of pretreatment, along with their general strengths and weaknesses.

6.7.1 Settling Basins

Larger stormwater recharge practices often include settling basins to filter out suspended sediments (Bardin et al., 2001; Pavelic et al., 2006; Yuan et al., 2019). Settling basins not only reduce the potential for clogging in downgradient EAR systems, but they can also provide a convenient location to remove accumulated sediment from the system. Settling basins can include dry (periodically inundated) or wet versions, which are typically referred to as detention basins. Settling basins can require more land than filtration or green infrastructure pre-treatment, but the practices are common, relatively easy to construct, and can serve as public amenities (Pavelic et al., 2006; Vanderzalm et al., 2014b). Settling basin design is important, however, and must consider peak flow rates, sediment load, and sediment composition.

Sediment composition can affect the performance of any pretreatment system. Settling basins, particularly if undersized, may only reduce larger sediment, leaving finer fractions to pass through to the infiltration basin or well. This is especially true in systems receiving runoff from agricultural areas, which may export large amounts of eroded sediments during tilling and planting seasons (Beganskas and Fisher, 2017). Pretreatment of sediment may also be particularly effective at removing pathogens. For example, sedimentation practices may remove pathogens bound to particles (Pitt et al., 2003).

A properly sized sedimentation basin, particularly as part of a larger pretreatment system, can be an effective way to maintain operation of any stormwater recharge practice. Pavelic et al. (2006) describe a pilot ASR system that uses a series of three detention basins to remove sediment and improve water quality before injection, in addition to providing an aesthetic amenity to the surrounding agricultural area. The series of basins is generally capable of keeping TSS to 50–200 mg/L, though wet weather peaks sometimes exceed this range (Pavelic et al., 2006). In the Parafield system in Australia, an in-stream detention basin serves to settle larger sediments and associated pollutants, after which a reedbed provides further treatment of the water before injection (Vanderzalm et al., 2014b). Maintenance plans should account for the possibility that when solids accumulate in settling basins they may concentrate heavy metals and other pollutants associated with the solids (Yousef et al., 1990).

6.7.2 Constructed Wetlands

Constructed wetlands are used for a variety of water treatment applications and can be an effective pretreatment option for stormwater (Hamadeh et al., 2014; Lazareva, 2010; Page et al., 2010a, 2010b, 2010c; Rousseau et al., 2008). Constructed wetlands can be divided into a few different categories. There are surface-flow constructed wetlands and subsurface-flow constructed wetlands (Hamadeh et al., 2014; Rousseau et al., 2008). There are also horizontal and vertical constructed wetlands, which refers to the water flow direction (Ghermandi et al., 2007; Kadlec and Wallace, 2008; Rousseau et al., 2008). In addition to pretreatment, constructed wetlands themselves can recharge stormwater into the aquifer. When constructed wetlands are designed for aquifer recharge, they are referred to as “leaky wetlands” (Maliva, 2020).

Constructed wetlands are effective in reducing suspended solids, nutrients, and organic carbon concentrations (Clark et al., 2015; Dillon et al., 2014; Kadlec and Wallace, 2008; Maliva, 2020). Constructed wetlands can also remove trace pollutants commonly found in stormwater such as heavy metals and can even provide some measure of pathogen treatment (Arden and Ma, 2018; Ghermandi et al., 2007; Kadlec and Wallace, 2008; Maliva, 2020). However, constructed wetlands do have a few disadvantages. They are subject to clogging and poor removal of pollutants when heavily loaded (Lin et al., 2006; Rousseau et al., 2008). Although clogging is a notable problem for many EAR systems, clogging of subsurface-flow constructed wetlands can be particularly problematic (Rousseau et al., 2008).

Wetlands as a pretreatment technology may remove pathogens (Page et al., 2012). Specifically, constructed wetlands may be an effective option for removing bacteria, but is not likely to be one for removing viruses and protozoa (Sidhu et al., 2010). Within a constructed wetland, emergent vegetation may work to enhance sedimentation, which can sequester pathogens associated with the settled particles. In addition, pathogens can adhere to vegetation or be inactivated in the root zone (Sidhu et al., 2010). In a study of a constructed reedbed (a type of wetland), a retention time of six days resulted in one $\log_{10}$ removal for bacteria (Sidhu et al., 2010). The authors concluded that the presence of enteric bacteria in recovered water was unlikely. However, in the same study of constructed reedbeds, the system needed more than 33 days of retention time to achieve one $\log_{10}$ reduction time for adenovirus and *Cryptosporidium.* The authors concluded that reedbeds cannot effectively remove adenovirus or *Cryptosporidium.*

Another design challenge of constructed wetlands in urban areas is that they need large areas of land, although subsurface flow constructed wetlands typically need less area than surface-flow construction wetlands. In constructed wetlands, the ideal conditions for pollutant removal are slow flow through shallow water and dense vegetation (Maliva, 2020). Such conditions require the constructed wetland to be sufficiently large. Removal percentages in constructed wetlands may also vary over the course of a wetland's life. Removal percentages depend mainly on temperature, residence time, seasonality (due in part to the effect of vegetation) and loading rate. Low temperatures can inhibit denitrification and therefore nitrogen removal rates. High peak flows can reduce solids removal rates (Rousseau et al., 2008).

Although constructed wetlands do not need much maintenance, at least compared with systems that might require pumps and electrical equipment, they do need some. Insufficient maintenance can lead to uneven flow and overloading (i.e., insufficient solids removal) in some parts of the system. Odors are a concern, especially in high-loaded systems with anaerobic conditions (Rousseau et al., 2008).

6.7.3 Green Infrastructure

Constructed wetlands can provide substantial water quality treatment but typically need large areas of land, which in many urban areas is not available. Green infrastructure practices offer many of the same benefits of constructed wetlands—sediment filtering, nutrient and other pollutant removal, aesthetic appeal—while requiring a fraction of the space. Here, we discuss several green infrastructure practices that can be used as pretreatment for stormwater recharge. For a thorough discussion of green infrastructure and its effect on groundwater quality, see U.S. EPA (2018b).

Biofilters, which are variably saturated and often incorporate facultative wetland vegetation, are more space efficient and can be implemented in a more decentralized way than wetlands for treating stormwater for EAR (Kerrigan et al., 2014); they can provide effective pretreatment (Le Coustumer et al., 2009; Macnamara and Derry, 2017; Shen et al., 2020; Zhang et al., 2014a, 2014b). Biofilters are vegetated soil filtration systems designed to remove sediment and nutrients from stormwater. Pollutants are removed through sedimentation, filtration, sorption, and biological uptake. A saturated zone can be incorporated to provide anaerobic conditions necessary for denitrification (Kerrigan et al., 2014). To target certain pollutants like metals and nutrients, various amendments can be added to the soil that promote enhanced sorption and filtering (Hirschman et al., 2017; Payne et al., 2019) Tree filter systems can be good candidates for these media amendments as well (Schifman et al., 2016).

Vegetation also serves a pretreatment role in swales. Swales are shallow open channels that are designed to treat stormwater runoff from adjacent impervious surfaces. They function by reducing particulate pollutants through settling and filtration. Vegetative uptake and adsorption also act to reduce dissolved pollutant concentrations, although a swale's main purpose is to reduce particulate pollution. A swale should be designed to convey, but not store, the peak discharge of the design storm and should be large enough to retain the flow from small storms (Maliva, 2020).

Although it is not a traditional pretreatment technology, full-depth permeable pavement can be used to treat stormwater before it enters an EAR system or before it simply infiltrates into the ground beneath the permeable pavement. Permeable pavement is generally more expensive than other green infrastructure, but the cost can be justified in high-density urban areas that lack space for a standalone treatment system and also have a demand for paved parking areas. Most full-depth permeable pavement systems include layers of gravel, geotextile, or sand beneath a permeable asphalt topcoat and are designed to capture the average design storm. Typically, the water is stored and then infiltrates into the groundwater. In this sense, permeable pavement systems act as a component of recharge operations. However, storms may be too large for the permeable pavement system to capture completely. Even when there is overflow (i.e., not all stormwater is captured), the discharged overflow can be cleaner than surface runoff from impermeable pavement (Kayhanian et al., 2019).

Clogging of permeable pavement by sand in heavy traffic areas can prevent proper functioning, as discussed in Section 6.7.1, and removal of anions and nutrients may be limited (Boving et al., 2008; Kayhanian et al., 2019). Permeable pavement systems remove particulate pollution through filtration and adsorption (Drake et al., 2013; Kayhanian et al., 2019; Scholz, 2013). Organic pollutants are degraded through microbial activity in the pavement system (Drake et al., 2013; Imran et al., 2013; Kayhanian et al., 2019; Scholz, 2013). Pervious concrete can also raise the pH of infiltrated water, thereby reducing the solubility of metals and protecting groundwater quality (Imran et al., 2013; Kayhanian et al., 2019). Permeable pavement may also reduce the temperature of infiltrating stormwater compared to runoff from traditional pavement (Drake et al., 2013).

6.7.4 Media Filtration

Media filtration generally refers to a range of physical filtration practices, ranging from coarse gravel filters to sand filtration to filter material with pore sizes on the order of micrometers. Owing to its ability to effectively remove sediment, media filtration is often used at injection wells because it can effectively remove sediment, even a small amount of which can clog a well screen (Bouwer et al., 2008; Lin et al., 2006; Pavelic et al., 2007). The selection of an appropriate media filtration system is generally dictated by the end use of the treated stormwater and the economics of the project (Maliva, 2020). Roughing filters are appropriate for passive systems where minimal maintenance is a priority and coarse sediment is the target pollutant. Granular-media filters, which can be placed in series with roughing filters, provide a higher degree of sediment removal and can also remove some pathogens and organics, but generally require more maintenance. Advanced media filtration is used to target dissolved pollutants and is rarely used to treat stormwater given its generally high cost and maintenance requirements.

6.7.5 Roughing Filters

Roughing filters are a pretreatment technology used to reduce suspended solids and turbidity in stormwater runoff (Maliva, 2020; Wegelin, 1996). Roughing filters generally consist of tanks or basins filled with gravel. They work by slowing flow and lessening the effective settling distance of a particle, i.e., rather than having to fall a distance on the order of feet, as would be the case in a settling basin, particles only have to fall a distance comparable to the pore gap size in the gravel. Still, given their rather large pore size (i.e., gravel), roughing filters are often used in series with other media filtration, such as sand filters, for a stepwise sediment removal process (e.g., Lin et al., 2006). In addition to particulate pollution, roughing filters can remove chemical contaminants and pathogens that are attached to suspended solids, although their main purpose is not to remove such contaminants.

When used prior to sand filtration, Maliva (2020) suggests, the basic goal of roughing filtration is to reduce turbidity to between 10 and 20 nephelometric turbidity units and TSS to between 2 and 5 mg/L. Given these effluent guidelines, system design therefore depends on incoming flow rates as well as turbidity and TSS concentrations of influent. Depending on these variables, studies have found roughing filters to reduce turbidity by 75–96% and TSS by 55–96%, with better performance generally being achieved by systems with smaller pore sizes (Collins et al., 1994; Lin et al., 2006; Wegelin, 1996). (The range of TSS reduction is so large because it depends on properties of the influent.)

6.7.6 Granular-Media Filters

Granular-media filters are mostly used to remove suspended solids from stormwater runoff before it enters the EAR system (Barry et al., 2017; Maliva, 2020; Segismundo et al., 2017; Zarezadeh et al., 2018), although sand filters have also been used as parts of treatment trains to remove dissolved organic carbon (Linlin et al., 2011). There are several types of granular-media filters, for example rapid-sand filters, rapid-pressure filters, and slow-sand filters (SSFs).

Although such filters may remove some particulate pollution, sand filters may not be effective at removing suspended solids to a level that is protective of well clogging. For example, an ASR pilot project in Adelaide, Australia (Urrbrae Wetlands Park), which injected stormwater treated with a constructed wetland followed by a rapid sand filter, failed after just six weeks owing to high suspended material, colloidal material, and elevated total organic carbon that was not effectively removed by the sand filter pretreatment system (Bouwer et al., 2008; Lin et al., 2006).

While rapid-sand filters and rapid-pressure filters are designed only to remove particulates, SSFs are used to remove suspended solids *and* pathogens (Maliva, 2020). In fact, SSFs can achieve a 2–4 $\log_{10}$ total coliform removal (Hendricks, 1991). Infiltration basins essentially act as SSFs. The main drawback of SSFs is that they need a larger area than rapid-sand filters and rapid-pressure filters. SSFs are also subject to clogging. If the water has high turbidity, the stormwater should pass through a roughing filter before entering the SSF (Maliva, 2020).

Instead of sand, some systems may use composite materials—sometimes called geomedia—to pretreat stormwater. Such geomedia may better avoid clogging than sand filters and may need replacement or regeneration only every 20 to 30 years (Ray et al., 2019). However, geomedia are certainly less used than sand. In fact, Ray et al. (2019) did not observe geomedia in field conditions, only in column studies.

6.7.7 Advanced Media Filters

A number of advanced media filters, such as membrane filters or ion exchange resins, can be used for stormwater pretreatment and can provide extremely high levels of treatment (Maliva, 2020). These systems tend to be costly, though, and are more commonly used for polishing of treated wastewater owing to the higher treatment requirements for wastewater (Bouwer et al., 2008; Maliva, 2020; Yuan et al., 2016, 2019). We discuss them here because of stormwater EAR's potential to become more widespread, and the corresponding need that greater levels of treatment be implemented—be it for maintenance of flow rates, water quality protection or human health protection.

6.7.8 Chemical Pretreatment and Combined Pretreatment Systems

Chemical pretreatment is often used to avoid clogging and protect water quality in EAR systems, particularly those systems that recharge water intended for subsequent recovery. Chemical pretreatment approaches that have been applied to recharge systems include disinfectants, pH adjustment, dissolved oxygen removal, and iron and manganese management (Maliva, 2020). Coagulation can also be used to manage recharge water properties. While most chemical treatments focus on the managing the chemical properties of the recharged water, some chemical pretreatment methodologies directly target pathogens in the recharge water. In this section, we discuss how chemical pretreatment has been used for recharge systems. Most chemical pretreatment examples are from ASR systems, given the need for high-quality recovered water. In most cases, however, if stormwater is to be recharged with the intention of subsequent recovery for drinking water or other purposes, many of the best practices below are still applicable.

For injection systems in carbonate or karst aquifers, pH management through pH reduction may be necessary to limit calcium carbonate precipitation, which can clog wells and limit recharge and recovery flow rates (Maliva, 2020). Decreases in pH can be achieved by supplementing the source of recharge water flow with an acid feed. Carbonic acid is commonly used for pH adjustment, as it presents fewer safety concerns than some other acids (such as hydrochloric acid or sulfuric acid). Carbonic acid has been used for pH adjustment at ASR wells in Florida, which has many karst aquifers. Alternatively, pH levels that are too low may cause corrosion of metal within the well or mobilization of metals in the aquifer, as discussed in Section 5.5 (Antoniou et al., 2012; Bouwer et al., 2008; Ibison et al., 1995; Pyne, 2005). Moreover, stormwater can have a naturally low pH compared to surface water or reclaimed wastewater and can often be low enough to maintain sufficient aquifer permeability and porosity through matrix dissolution (see discussion in Section 4.2.2). In all cases, site-specific aquifer and water quality characterization is critical to determine the optimum pH balance.

Injecting water that has a high dissolved oxygen content into an aquifer can result in oxidation of certain minerals and subsequent mobilization of harmful metals. Pyrite (FeS_2), arsenopyrite ($FeAsS$), and other sulfide minerals that are stabile in naturally anoxic aquifers are particularly susceptible to oxidation. This can result in the release of iron, sulfate, and associated trace constituents such as arsenic, cobalt, nickel, and zinc into groundwater (Arthur et al., 2005; Bouwer, 2002; Mirecki, 2006). The oxidation process also lowers ambient pH, which can result in mobilization of iron and manganese from carbonate minerals such as siderite (Antoniou et al., 2012; Ibison et al., 1995; Pyne, 2005). ASR systems have used dissolved oxygen removal to reduce the redox potential of the source of recharge water and control the leaching of arsenic, iron, and manganese into groundwater (Bell et al., 2009; Maliva, 2020). Uncatalyzed chemical reduction and volatilization are the two most common methods of dissolved oxygen removal for chemical pretreatment in ASR systems (Maliva, 2020). In uncatalyzed chemical reduction, reduced sulfur compounds, such as sulfide, sulfite, or thiosulfate, are added to the recharge water (Pearce and Waldron, 2011). This approach has the disadvantage of adding dissolved solids to the water. In volatilization, a carrier gas or negative pressure is used to strip dissolved oxygen out of the recharge water. A disadvantage of volatilization is that it may cause other, unwanted chemical changes to the recharged water.

An alternative approach to managing water quality impacts from oxidation is pre-oxidation, or the intentional inactivation of the reactive aquifer phases. Pre-oxidation is based on the observation of decreasing iron concentrations in recovered water over successive ASR cycles with oxic recharge water, suggesting the formation of stable, oxidized precipitates on mineral surfaces (Antoniou et al., 2012; Pyne, 2005). Pre-oxidation accelerates the natural oxidation process, often through chemical additions. Permanganate (MnO_4^-), a strong electron acceptor, has been added to recharge water and shown to deactivate pyrite. Permanganate also forms stable Mn-oxides that adsorb free Fe(II) and Mn(II) (Antoniou et al., 2014; Maliva, 2020). In laboratory studies, pyrite leaching has decreased by 63% after a permanganate treatment was used (Antoniou et al., 2014). Permanganate also has the potential to limit production of Mn(II) (Antoniou et al., 2014; Maliva, 2020).

Impacts from high suspended solids, nutrients, and pathogens can also be addressed through chemical addition. Abel et al. (2015) found that using aluminum sulfate and iron chloride for coagulation removed 65% of suspended solids in laboratory tests. In the same study, coagulation removed 80% of phosphorus and 16–22% of dissolved organic carbon from the recharge water. Coagulation also increased removal of *E. coli* from 2.5 $\log_{10}$ removal to 3.8 $\log_{10}$ removal and increased total coliform removal from 2.6 $\log_{10}$ removal to >4 $\log_{10}$ removal (Abel et al., 2015). Sasidharan et al. (2021b) also suggest the use of iron oxides as an in-situ soil treatment to increase virus attachment and solid phase inactivation.

There is evidence of some stormwater BMPs using chemical agents, such as an organosilane derivative (C-18 organosilane quaternary), to inactivate FIB (ASCE, 2014). Researchers in Australia have studied the effects of the various pretreatment methods and the effectiveness of these methods (Petterson et al., 2016):

- Residential stormwater flowed through a grass swale, underlain by a bioretention trench or central sand filter zone. The stormwater then flowed to a UST that consisted of many small cells enclosed by a geotextile. In this scenario, pretreatment on average produced a 0.61 $\log_{10}$ removal of *E. coli*. The authors noted that the grass swale filter was likely not a significant source of pathogen reduction. Within the storage tank, pathogen reduction likely occurred through sedimentation and dark inactivation.
- Stormwater flowed from a multi-story parking garage, road, and grassed sports field into a treatment train that consisted of sedimentation tanks, a biofilter, and a storage pond. In this scenario, pretreatment produced an overall 0.32 $\log_{10}$ removal of *E. coli,* although the storage pond may have increased *E. coli* concentrations.
- Stormwater flowed from a predominantly residential area into a treatment train that consisted of a sedimentation basin, a wetland, an open storage pond, and UV disinfection. In this scenario, pretreatment produced a less than one $\log_{10}$ reduction for *E. coli,* just over a one $\log_{10}$ reduction for *Clostridium perfringens,* and approximately a 0.5 $\log_{10}$ reduction of somatic coliphages.

Additionally, pretreatment systems may intentionally expose stormwater to sunlight, which can inactivate FIB (ASCE, 2014; Page et al., 2012). The pH of the influent stormwater is relevant, as low and high pH can inactivate bacteria in stormwater (ASCE, 2014). If the goal of an EAR project is to eventually recover recharged stormwater, multi-barrier treatment trains may be used to treat urban stormwater via constructed wetlands before recharge and treat recovered water via chlorination and UV radiation, as research has demonstrated 1.4, 2.6, and >6.0 $\log_{10}$ removals for rotavirus, *Cryptosporidium*, and *Campylobacter*, respectively, for this technology (Page et al., 2010c). Pretreatment that removes nutrients will not necessarily reduce pathogen concentrations, as research on the association between nutrients and pathogens is mixed (ASCE, 2014).

Bioretention ponds, sand filters, and wet retention ponds may be able to reduce FIB to some extent (ASCE, 2014). However, grass strips and swales do not reduce FIB concentrations (ASCE, 2014). Ahmed et al. (2019) provides pathogen $\log_{10}$ reduction values associated with pretreatment technologies such as retention ponds, constructed wetlands, and biofilters.

6.8 EAR OPERATIONS AND MAINTENANCE

Proper operation and maintenance of an EAR system is critical to protecting groundwater quality and ensuring the sustainability of the system. Because the types of EAR systems using stormwater are diverse, this section summarizes some common best practices for two overarching types of recharge practices: infiltration practices and injection wells. For all practices, local codes will dictate specific operation and maintenance practices such as recommended inspection and cleaning intervals, fencing requirements for safety purposes, and more. More detail on planning, design, construction, operation, monitoring, and closure of such projects is available in the *Standard Guidelines for Managed Aquifer Recharge* (ASCE, 2020). Also, an overview of common BMPs, including discussion of best operations and maintenance practices, can be found at EPA's national menu of BMPs for stormwater (U.S. EPA, 2020).

6.8.1 Infiltration Practices

Infiltration Basins

Infiltration basins are typically the most cost-effective EAR method and are the easiest to maintain (Jeong et al., 2018). There are several options for avoiding clogging in infiltration basins, including repeated drying and cleaning cycles, as well as various methods of pretreatment. Pretreatment is sometimes determined to be more economical than repeated drying and cleaning cycles. Testing may be necessary to optimize operations at any given site (Bouwer, 1988). Chemical contaminants and nutrients can be removed in infiltration basins using engineered geomedia (Grebel et al., 2016; Hirschman et al., 2017; Ray et al., 2019; Spahr et al., 2020). Such geomedia can be added to soil layers in smaller green infrastructure systems or to the surficial sediments of larger infiltration basins (Ray et al., 2019).

When managing a stormwater infiltration basin, operators should consider the tradeoffs between maximizing the infiltration rate and minimizing the clogging rate. They may be tempted to direct high flows of stormwater to an infiltration basin, but these higher flows may increase turbulence and therefore the amount of suspended solids and the rate of clogging (e.g., see discussion in Section 6.9.2). Onsite testing will help determine how to manage the tradeoffs (Bouwer, 1988).

In addition to managing the flow of influent water, personnel may want to manage the depth of pooling water in a stormwater infiltration basin. Even though higher water levels create a greater hydraulic head and a higher basin storage capacity, they can cause compaction of the clogging layer and may decrease the infiltration rate (Bouwer, 1988; Maliva, 2020). Higher water levels also may be associated with a decrease in the turnover rate of the basin, thereby promoting algae growth and clogging (Bouwer, 1988; Bouwer et al., 2008). Algae can also uptake carbon dioxide, increase the pH of the water, and cause calcium carbonate to precipitate and further clog the basin (Bouwer, 1988; Bouwer et al., 2008; Fernandez Escalante, 2015; Heilweil and Marston, 2011; Schuh, 1990). Shallow basins may also be convenient because they allow for rapid draining and drying of the basin (Bouwer et al., 2008).

Personnel managing stormwater infiltration basins may also consider controlling the influent water temperature to avoid gas bubbling (Pedretti et al., 2012), although this may be difficult. Basins should be lined with vegetation to minimize bank erosion by stormwater runoff (Bouwer et al., 2008; Maliva, 2020). Infiltration basin banks may also be lined with plastic or cement. Even properly operated and designed stormwater infiltration basins must undergo maintenance. Even if all physical and chemical clogging agents are removed, the growth of algae and autotrophic bacteria will make maintenance necessary. Dust may also be blown into the basin (Bouwer et al., 2008; Maliva, 2020). Basins must be regularly dried and cleaned to reduce clogging and maintain desired infiltration rates (Badin et al., 2011; Bouwer, 1988; Bouwer et al., 2008; Dutta et al., 2015; Ma and Spalding, 1997; Morrison et al., 2020; Pedretti et al., 2012; Regnery et al., 2020; Schuh, 1990). Soil clogging and infiltration capacity are difficult to model and estimate (Pedretti et al., 2012), and the optimal drying and cleaning schedules are site-specific (Bouwer, 1988).

Generally, if the clogging material is inorganic, best practices dictate that it be removed after the basin is dried (Bouwer, 1988; U.S. EPA, 2020). If the clogging material is organic, drying alone may be sufficient (Bouwer, 1988; Bouwer et al., 2008). Clogging material can often be removed mechanically with scrapers, front-end loaders, graders, or manually with rakes (Bouwer et al., 2008; Ma and Spalding, 1997). When removing the clogging layer, operators may also want to remove less permeable surface material to promote higher infiltration rates (Bouwer et al., 2008; Estragnat et al., 2018; Maliva, 2020).

After the drying period and removal of clogging material, infiltration basins are often disked to break up any compacted layers that may still exist (Bouwer et al., 2008). After disking, the basin bottom may need to be smoothed (Bouwer et al., 2008). Research has been conducted to identify how to balance pretreatment and cleaning and drying cycles. In one study, the researchers used five examples to identify the optimal balance of pretreatment and cleaning and drying cycles (Pedretti et al., 2012).

- After seven days of no maintenance, the infiltration basin reached 37% of its initial infiltration rate. After this first trial, a 37% reduction in infiltration rate became the benchmark for the following trials.
- With treatment addressing biological clogging only, the infiltration basin reached 37% of its initial infiltration rate after 9.5 days.

- With full treatment of physical clogging, the infiltration basin reached 37% of its initial infiltration rate after 28 days.
- With only 50% of physical clogging treated, the infiltration basin reached 37% of its initial infiltration rate after 18–20 days.
- With 80% of physical clogging treated, the infiltration basin reached 37% of its initial infiltration rate after 25–27 days.

Note that infiltration and clogging rates are site-specific. Although the effects of addressing biological and physical clogging seen in Pedretti et al. (2012) may not be translatable to all other infiltration basins, that study does show that treating biological and physical clogging can reduce maintenance needs. Pedretti et al. (2012) also demonstrate that the tradeoffs between maintenance and addressing clogging can be managed and optimized.

Because infiltration rates sometimes abruptly decrease (Racz et al., 2012), it is important to pay close attention to the operation and performance of a stormwater infiltration basin. If personnel do not want to implement the cleaning and drying cycles, the basin can be designed to use underwater robots to scrape the soil surface during infiltration (Pedretti et al., 2012). California's Orange County Water District uses automated basin cleaning vehicles to manage clogging (Bouwer et al., 2008). Alternatively, to avoid the necessity of repeated cleaning and drying cycles, creating a furrowed bottom (instead of a flat bottom) may promote infiltration in a basin; for maintenance, operators can create waves in the basin to resuspend and wash off sediment deposited on the slopes of the furrows (Maliva, 2020).

There is debate as to whether vegetation should be grown in stormwater infiltration basins. Vegetation may increase evapotranspiration (Maliva, 2020). It may also contribute to vector problems, clog the soil with roots, and interfere with the cleaning and disking of the basin (Bouwer et al., 2008); however, other sources indicate that roots of vegetation may enhance infiltration and uptake nutrients (Bartens et al., 2008; Gonzalez-Merchan et al., 2012; Maliva, 2020; Mindl et al., 2015). Regardless of the vegetation, care should be taken so that infiltration basins do not become breeding grounds for excessive insects (Bouwer, 1988). Over the years, operators may need to import new sand to replace sand removed along with clogging layers (Bouwer et al., 2008). Stormwater infiltration basin managers may also consider using disinfectants to control algal growth (Pedretti et al., 2012).

Infiltration Trenches

Like infiltration basins, infiltration trenches are subject to clogging. However, because they are so narrow (usually about 1 meter wide), they cannot be redeveloped, cleaned, or restored as easily as basins to restore infiltration rates (Bouwer et al., 2008; Maliva, 2020). This makes pretreatment especially important for infiltration trenches (Bouwer et al., 2008; Maliva, 2020). Pretreatment can be achieved in a portion of the trench itself by installing a small ditch block to create a settling chamber or by using a sand filter and possibly a geotextile filter fabric (Bouwer et al., 2008) or a fabric filter (Maliva, 2020). Vegetation around a stormwater infiltration trench can also filter coarse sediment and prevent erosion, further preventing clogging (Maliva, 2020).

As noted in the previous sections, it is particularly difficult to clean infiltration trenches. Yet pretreatment may also be an expensive option. In some cases, it may be more economical to regularly construct new infiltration trenches than to provide pretreatment (Bouwer et al., 2008; Maliva, 2020). A trench may need rehabilitation every 5–15 years (Maliva, 2020).

Stormwater BMPs

Capturing stormwater close to its sources will minimize contaminant and sediment concentrations in the runoff (Stephens et al., 2012) and will therefore decrease the need for any pretreatment. Previous surveys have indicated that infiltration BMPs have high failure rates from clogging, and that the clogging happens quickly. Proper operation and maintenance is therefore important. Personnel should consider geography and climate before constructing stormwater infiltration BMPs (Emerson and Traver, 2008). Simulation of infiltration practices should not be based on a single representation of the rate of infiltration, because that rate can change dramatically with the seasons and with variations in temperature (Braga et al., 2007; Emerson and Traver, 2008; Maxwell et al., 2003). Single field tests may not be sufficient for characterizing or designing a stormwater infiltration BMP (Emerson and Traver, 2008).

These BMPs' design, maintenance, and operation should ensure that the following are true for their entire lifetimes (Kayhanian et al., 2019):

- There is adequate reservoir capacity to capture stormwater/runoff.
- The surface pavement remains highly permeable and unclogged.
- The soil below the pavement remains permeable.

The use of road salts or other deicing materials can have a deleterious effect on infiltration practices through dispersion-based clogging. When mixed with soil, sodium—a constituent of common road salt—results in chemical dispersion of small soil particles such as clays, leading to mobilization of this finer fraction and clogging of infiltration media (Amrhein et al., 1992; Kakuturu and Clark, 2015).

Pervious pavement can be unclogged through routine maintenance by vacuuming (Kayhanian et al., 2019). General stormwater BMPs may only require periodic inspection for clogging, erosion, health of vegetation, and infiltration times after storms. Regular maintenance may include mowing of grass, trimming of vegetation, removal of unwanted vegetation, and collection of garbage (Maliva, 2020).

Dry Wells

Dry wells enable rapid infiltration of stormwater directly to the vadose zone, bypassing surficial soil layers that may not be suitable for infiltration. Although this design enables stormwater infiltration in areas that be otherwise unsuitable, it can lead to more rapid clogging or contamination issues depending on the dry well's hydrogeological setting and surrounding land use (Edwards et al., 2016; Geosyntec, 2020). Moreover, clogging can be difficult or impossible to fix once it occurs (Maliva, 2020; NRC, 2008; U.S. EPA, 2012). Still, in addition to proper siting and design, certain operation and maintenance measures can mitigate these risk factors.

Many dry well systems have some type of pretreatment, which may include a sedimentation basin, sedimentation chamber, oil-water separator, or absorbent pad (Maliva, 2020). Maintenance for settling basin or chamber pretreatments is similar to maintenance for standard infiltration basins, including periodic removal of trash, debris, or sediment. Maintenance for filter-based pretreatments includes periodic replacement of the filter media. Some dry wells are constructed with a gravel or sand filter at the surface to reduce any clogging of the underground well surfaces (Geosyntec, 2020; Maliva, 2020; Talebi and Pitt, 2014). If maintenance personnel note prolonged standing water in the surrounding settling basin, this surface filter may be clogged and should be replaced. Watershed-based prevention measures like street sweeping and routine inspections for actively eroding soils can also help reduce dry well clogging (Geosyntec, 2020).

Given the potential for dry wells to rapidly convey contaminated runoff to the subsurface, some states also recommend spill response plans and public outreach measures for dry wells, especially those in areas with surrounding, high-risk land uses. In its guidance and screening protocols for dry well siting, design and operation (Geosyntec, 2020), the state of California recommends the following:

- Designers should characterize the surrounding land use in terms of risk for contaminant spills (industrial land uses tend to pose the highest risk).
- Property owners of land with high-risk uses that contains dry wells should develop spill response plans.
- Local fire departments should be made aware of dry well locations to limit runoff of flame-retardant chemicals.
- Municipal construction departments should take extra precaution when granting approval to construction projects that drain to dry wells.

6.8.2 Injection Wells

Although some pretreatment options may depend on the stormwater characteristics and the aquifer characteristics, one pretreatment option is true across all injection wells. Before stormwater is injected, the water needs to be treated to remove almost all suspended solids (Bouwer, 1988). Sand or membrane filtration can be used to accomplish this (Bouwer et al., 2008; Dillon et al., 2001; Stuyfzand and Osma, 2019).

During the operation of stormwater injection wells, personnel should ensure that stormwater is injected through a relatively small pipe in the injection well and that the pipe ends below the water level (Bouwer, 1988). If the pipe ends above the water level, the stormwater will free fall into the wells and air will become entrained in it (Bouwer, 1988). Entrained air can reduce the hydraulic conductivity of the aquifer around the well and thus reduce the injection rate as well (Bouwer, 1988; Mizrahi et al., 2016). This consideration pertains both to the design and the operation of the stormwater injection well.

Stormwater injection well operators should manage the injection pressure used, because increased injection pressure is associated with clogging (Bouwer et al., 2008). Even when solids are removed, the injection well will still require periodic pumping, backflushing, or redevelopment of the well to prevent clogging (Bouwer, 1988; Bouwer et al., 2008; Dillon et al., 2016; Fernandez Escalante, 2015; Jeong et al., 2018; Zeng et al., 2019). In some cases, frequent pumping of stormwater injection wells may be more effective than pretreatment of the influent water (Bouwer et al., 2008). In a five-year ASR trial, Pavelic et al. (2006) showed that periodic (once or twice per year) redevelopment or reconstruction of an injection well by jetting with compressed air (a process referred to as "airlifting") was able to adequately restore injection rates. This success was unexpected, as the injectate was particularly high in solids, with average annual concentrations of TSS ranging from 29 to 169 mg/L.

6.9 RECHARGE VOLUME OPTIMIZATION

In addition to recharge water and site characteristics, the operation of an EAR system can have an important influence on the water volume able to be recharged. There are a range of sizes and types of systems, from small green infrastructure practices to regional ASR systems. Although very different in design and operation, each has its capacity to recharge stormwater that is a function of multiple factors. Each has its own set of limitations that can result in diminished performance. Each also has ways in which it can be optimized, ways that are unique to the intermittent nature of stormwater delivery. This section discusses those factors.

6.9.1 Alternative Sources of Recharge Water

Stormwater is often seasonal, which means stormwater recharge facilities may sit dormant for much of the year. Moreover, given common design standards for retention volume, infiltration basins tend to be runoff-limited instead of infiltration-limited (Beganskas and Fisher, 2017; Miller, 2006; O'Leary et al., 2012). To maximize aquifer recharge during dry times, some systems will divert other surface waters to stormwater infiltration basins (O'Leary et al., 2012), which not only better utilizes the infiltration system but reduces surface water losses to evapotranspiration.

As illustration of unused recharge capacity, a series of roughly 2,200 infiltration basins were constructed in Nassau, New York, beginning in the 1930s. Most basins were designed to accommodate runoff from 5 inches of rainfall (Aronson and Prill, 1977), a volume much higher than current stormwater design standards and one that leaves most basins underutilized. Accordingly, Aronson and Prill (1977) conducted a modeling study to determine the additional recharge capacity the basins could accommodate, as a function of design storm size, without overflowing. They found that under most conditions, recharge volumes could be increased by at least a factor of two using alternative sources of recharge water. In the years since the study, others have observed infiltration rates in older basins to drop dramatically—indicative of natural clogging over time (Bouwer et al., 2008), and suggesting that the window for enhanced recharge with alternative sources of recharge water may be finite.

6.9.2 Infiltration Basin Loading Rate

Basin infiltration rates tend to decrease over time as more sediment is delivered to the system and surface soils begin to clog, though trends vary depending on basin soils and the quality of the stormwater being delivered to the system (Beganskas and Fisher, 2017; Bouwer et al., 2001; Racz et al., 2012). Racz et al (2012) measured infiltration rates of a 7-acre infiltration pond over two years and documented decreases in infiltration rates. Infiltration rates were initially greater than 3 feet/day and correlated with pond stage, but over the wet season decreased to 1 foot/day and became decoupled from pond stage, indicating a limitation of hydraulic conductivity as opposed to static head. By the end of the season, the infiltration had dropped to a minimum of 4 inches/day. Pond managers were however able to restore the initially high infiltration rates by scraping the accumulated fine-grained sediment from the bottom of the pond at the end of the wet season and exposing the native, coarser pond bottom. In comparison, infiltration rates in the 4.3-acre infiltration basin evaluated by Beganskas and Fisher (2017) showed no decline over six years, though basin managers periodically removed accumulated sediment. Also, and perhaps more importantly, the basin included a higher, sandy portion that was only accessed during large storms. Basin infiltration rates in the majority of the pond system may have declined over time, but these declines were likely offset by the sandy, underutilized portion of the system.

6.9.3 Injection Well Pumping Rate

As with infiltration basins, the rate at which a well will clog is generally proportional to the amount of water that passes through the well, though the rate also depends on site-specific conditions such as water quality and aquifer characteristics (Dillon and Pavelic, 1996; Dillon et al., 2014; Lin et al., 2006; Pavelic et al., 2006, 2007). From a management perspective, the amount of water injected is a function of the flow rate and the duration of pumping.

In a study designed to evaluate the effect of pumping cycle duration (injection plus recovery) and water quality on injection rates and recovery efficiencies, Page et al. (2011) observed no reduction in injection rate or recovery efficiency when injecting potable water for cycles ranging from less than one day to 90 days. When injecting stormwater into the same well using a pumping cycle of 67 days, they observed a 20% reduction in injection rate.

In a similar study of the Andrews farm ASR system, Pavelic et al. (2006) evaluated the reasons for clogging over five years. Although they expected significantly more clogging of the well given the membrane filtration index (MFI) value of the injectate (yearly TSS averages of 29–169 mg/L and MFI of 400–2,600 s/L^2) they found that calcite dissolution of the aquifer at least partially offset well screen clogging, with further offsets realized through routine well redevelopments that occurred after injection of 11 million gallons (66 million gallons were injected over four years). MFI is a standardized test of the rate at which water clogs a membrane filter; higher MFI values generally mean higher potential for clogging. Moreover, they observed that the extended well redevelopment that took place at the end of every year fully restored the specific capacity of the ASR well to its initial level.

6.9.4 EAR System Modeling

Simulation modeling is a useful and common approach used for planning and design of EAR systems (Ringleb et al., 2016), including characterization of risk. Several authors have modeled stormwater EAR using a variety of approaches. Although each study had unique objectives, the following provides an overview of their general objectives, approaches, and modeling programs:

- Newcomer et al. (2014) used a HYDRUS-2D model and long-term water budget to quantify urban recharge rates, volumes, and efficiency beneath an infiltration trench and an irrigated lawn. Combined with in situ measurements, they found the methods to be complementary and useful for simulating historic and future recharge conditions.
- Sasidharan et al. used HYDRUS 2D/3D software to characterize infiltration dynamics of dry wells in heterogeneous soils (Sasidharan et al., 2019), dry wells under constant head conditions (Sasidharan et al., 2020), and dry wells and infiltration basins (Sasidharan et al., 2021a) under a range of system configurations and hydrogeologic parameter values. Their work helps point to conditions that are optimal for drywell implementation and helps to characterize subsurface infiltration dynamics more precisely.
- Thomas and Vogel (2012) used a multivariate regression model to quantify the effects of a groundwater overlay district in Boston's Back Bay neighborhood on local groundwater tables. Using explanatory variables that included rainfall, potential evapotranspiration, previous groundwater elevations, and the location and capacity of installed stormwater recharge BMPs, the model showed small but statistically significant positive effects of the overlay district on groundwater levels.
- Miller (2006) took a mechanistic forensic approach, using groundwater mound and soil characteristics to back-calculate historical infiltration rates. This study used the UNSAT-H surface infiltration model, which is based on a version of the Richards equation, to model seepage of the recharge basins based on system design, climate, soil type, and vegetation. This was then coupled with MODFLOW-SURFACT to calculate the recharge from the infiltrated water and groundwater mound characteristics from monitoring well data.
- Clark et al. (2015) used the Water Community Resource Evaluation and Simulation System (WaterCress) model to simulate runoff, recharge, and recovery of a 3,930-acre catchment using ASR wells for stormwater harvesting and recovery under a range of development and climate scenarios.
- Talebi and Pitt (2014) used level-logger data of infiltration events for dry wells in New Jersey to calculate infiltration rate equation parameters for Horton and Green-Ampt infiltration equations. Modeling was performed to investigate the effectiveness of onsite dry wells to limit stormwater flows into the local drainage system.
- TCEQ (2020) developed the Texas Aquifer Storage & Recovery Applet (TxASR app) to determine the recoverability for a single ASR well assuming a homogeneous aquifer and steady flow conditions. The TxASR app uses an interactive dashboard that allows users to define values for parameters that characterize the aquifer, pumping flow rates, and pumping schedule. It produces estimates of recovery efficiency and pumping time as well as graphs of important operational and efficiency parameters.

The authors of each study stress the need for site-specific parameters and field validation of approaches, as comparisons to similar studies in the literature showed wide variability. Talebi and Pitt (2014) found values for infiltration rate parameters to be orders of magnitude different than what is observed in the field. Newcomer et al. (2014), in estimating the recharge efficiency of their study practice (calculated as 58–79%), found that comparable studies estimated efficiencies of 40–99%, a range that provides for a very wide margin of error if applied to individual sites.

7 KNOWLEDGE GAPS

In the course of this review several knowledge gaps were identified where additional research could help advance effective and safe stormwater EAR. The following list is not exhaustive, but highlights some of the areas in which the need for information is the greatest.

Where in the United States are site physical conditions suitable to EAR?

Research is needed into where in the nation, and why, conditions are such that stormwater EAR may be most effective. Although stormwater recharge is a natural process and an ancillary part of many existing stormwater management approaches, EAR using stormwater as a primary objective is still relatively uncommon. As evidenced by the studies described in Section 4 and Table 4-2, the range of hydrogeologic conditions, design approaches, and scale is vast, making it difficult to draw consistent conclusions about what design approaches are more optimal than others. As stormwater EAR systems grow in number, we will learn more about infrastructure performance in diverse subsurface systems, under a myriad of land use contexts.

Modeling studies will be helpful in identifying the optimal areas for infiltration via EAR systems. Progress has been made in advancing site suitability analyses, tools, and maps for ASR, e.g., the TX ASR and AR suitability map developed by Shaw et al. (2020) and discussed in Section 6.1.1, which could be adapted to stormwater EAR. To increase the probability that suitable sites will be selected and that new stormwater EAR projects will succeed, there is a need for new data, tools, and models to characterize infrastructure performance, including information on maintenance that will reduce clogging of EAR systems.

What is the potential for stormwater EAR in the United States from a regional water supply perspective?

Urbanization, largely through the increase of impervious surfaces, alters natural water balances by increasing surface runoff and decreasing natural aquifer recharge. Combined with additional water supply demands of those urban populations, the effects of urbanization and population growth have depleted many aquifers across the United States. Additionally, urban runoff creates water quality problems, as it exports pollutants from the urban surfaces and creates unnatural hydrologic regimes in downstream waterbodies, causing a range of impacts.

Stormwater EAR provides an opportunity to simultaneously address both water quantity and water quality imbalances caused by widespread urbanization. As noted in Section 4, a few municipalities have realized (e.g., Chandler, Arizona; Nassau County, New York) or are realizing (e.g., Los Angeles, California) these dual benefits, but these areas represent a very small portion of the United States. To better understand the regional potential for stormwater EAR to restore depleted aquifers, more research is needed into the degree of aquifer depletion and the volume of available stormwater. Numerical modeling studies of stormwater flows would be helpful for estimating the water volumes available for recharge.

Studies assessing the long-term performance of EAR systems, specifically as it relates to site conditions, EAR maintenance, and system monitoring, would be helpful in guiding the design and operation of new systems. Such performance evaluations should evaluate the effects of changing environmental conditions, including climate change and land use change, on EAR system infrastructure and operation.

What are the greatest risks associated with more widespread adoption of stormwater EAR?

Stormwater can carry a wide range of pollutants, including nutrients, metals, pathogens, and trace organics. The concentrations of these pollutants vary according to the presence or absence of their sources in the watershed, and their potential impacts vary in terms of acuity and persistence. Given the variability of these pollutants in stormwater, the enormous volume of regional aquifers, and the limited number of existing stormwater EAR systems, limited information exists to quantify existing impacts and predict future impacts. Importantly, as stormwater EAR adoption becomes more widespread, the potential magnitude of these impacts is likely to increase.

In evaluating risks posed by a myriad of contaminants, it would be helpful to develop frameworks for subsurface geochemical analysis, including processes occurring in the aquifer and in the vadose zone. Nutrient attenuation in EAR systems would be a fruitful area for further research, given both anticipated short-term impacts (after transport through shallow aquifers) and the potential for long-term impacts to ecosystems and drinking water systems that use deep aquifers. Below, we discuss pathogens and trace organics, which present a number of challenges owing to uncertainties in their presence, fate and transport, and persistence.

Pathogen Risks

Stormwater contains varying amounts of pathogenic microorganisms, including bacteria, protozoa, and viruses, that can have detrimental human health effects. Although there are processes in the subsurface that result in inactivation or attenuation of such pathogens after recharge occurs (Page et al., 2010c, 2015a), these mechanisms are not fully understood. Some subsurface conditions—such as the lack of UV light, lower temperatures, and saturated soils—may even be conducive to pathogenic persistence or growth (ASCE, 2014). Where stormwater EAR projects are intended for subsequent recovery and use of the recharged water, more information is needed on potential human health impacts from the recovered water (Maliva, 2020; Zhang et al., 2013).

In particular, field studies are needed to investigate the survival and attenuation of pathogens in EAR systems. Given the wide variability of potential pathogen sources and the complexity associated with fate and transport processes, risk assessments related to pathogen fate and transport associated with EAR using stormwater may need to be site-specific. Results from laboratory studies on the fate and transport of pathogens in soil and aquifers can be misleading, as it is extremely difficult to replicate the complexity of these substrates (Page et al., 2015a; Sidhu and Toze, 2012). Promising results have been obtained from a very small number of field studies (e.g., Page et al., 2010c), but these results may not apply to other sites given the variability between sites. It will be important to identify those site-specific characteristics that support inactivation of pathogens.

Organic Compound Risks

Similar to pathogens, organic compounds like pesticides and PAHs can cause human health and environmental impacts. Also, identification of their sources and transport and degradation mechanisms are complex, variable, and in some cases unknown. PAHs have been detected at concentrations as high as 3500 mg/kg in runoff from parking lots with sealcoat. Parking lot sealants or sealcoats are used extensively in the United States to protect asphalt and enhance appearances. Although automobile exhaust, lubricating oils, gasoline, and tire particles are sources of PAHs in urban environment, parking lot sealcoat may be the primary source of PAHs in urban runoff (Mahler et al., 2005). Unlike pathogens, some organic compounds or their degradation byproducts can persist for decades. Traces of chlordane can still be detected in some areas, even though that pesticide was banned in the 1980s (Pitt et al., 1994); more recently, PFAS compounds are being detected in soils and stormwater (U.S. EPA, 2018b; Xiao, 2012). Further study is needed on the fate and transport of PFAS and other trace organic contaminants in diverse hydrogeologic settings. Although these incredibly persistent compounds are not being widely detected in aquifers, more widespread adoption of stormwater EAR may inevitably lead to increasing concentrations unless advances in detection and pretreatment are made.

Additional studies would be useful for identifying probable organic contaminants in stormwater within a given watershed based on land use within that watershed. This would allow for targeted monitoring and pretreatment of organic chemical contaminants, based on land use and expected occurrence.

A range of other trace organic compounds can be found in stormwater, including pharmaceuticals, antibiotics, synthetic and natural hormones, personal care products, detergent metabolites, antimicrobial agents, brominated flame retardants, perfluorooctane surfactants, fragrance and flavoring compounds, insect repellants, and x-ray contrast agents. Few studies have quantified the myriad toxic trace organic compounds potentially present in stormwater. It is difficult to determine if the results of these studies are characteristic of organic chemical occurrence in urban areas nationally, or to draw consistent conclusions that may link certain indicators (e.g., land use type, length of roadways) to their presence. More research is needed not only to determine which compounds may be present in stormwater, but also to prioritize which compounds pose greater risks to humans and the environment.

Are there locational or design factors for EAR of stormwater that suggest increased risk, particularly risk of water quality contamination?

To address risks to groundwater quality, new and improved data, tools, and models that characterize risks to aquifers are needed, as well as information on how such risks can be reduced with proper siting of EAR projects, pretreatment, system maintenance, and other means (e.g., Alan Plummer Associates, Inc. et al., 2010; Geosyntec, 2020). Frameworks can be developed for characterizing risks from EAR operations based in part on anticipated end-uses of the groundwater (e.g., drinking water, mitigation of salt-water intrusion, dilution of brackish aquifers, or eco-restoration).

Risk factors for water quality contamination associated with stormwater EAR are variable and can be largely site-specific. Urban land uses tend to contribute greater amounts of nutrients and pathogens (Ahmed et al., 2019), industrial land uses can be hotspots for chemical pollutants (Geosyntec, 2020), gas stations are associated with petrochemical contaminants (Borden et al., 2002), and metals and PAHs are broadly associated with roadways and vehicle traffic (Masoner et al., 2019; Pitt et al., 1994; Song et al., 2019; Weiss et al., 2008). Stormwater quality needs to be better characterized under a range of development and land use settings. Such studies should include analyses of temporal variability in stormwater quality, including changes in stormwater quality (e.g., increased pollutant concentrations during first flush) seasonally and during runoff events, that can help inform the design of infrastructure. Still, these associations are broad: site-specific information is needed for proper source identification and control.

Risk factors for water quality contamination also depend on contaminant transport pathways and the presence of potential contaminant receptors. For example, surficial recharge practices receiving contaminated runoff may pose a risk to domestic supply wells which tend to be shallower than larger municipal supply wells. Similarly, the siting and design of dry wells and injection wells that recharge into deeper aquifers must account for the potential presence of municipal supply wells, as contamination of these aquifers could lead to broad human health impacts. In all cases, proper screening of contaminant presence, transport pathways and potential receptors is needed to reduce water quality risk factors.

Water quality differences between infiltrated stormwater and ambient water in an aquifer can result in mobilization of subsurface contaminants (Dallman and Spongberg, 2012; Maliva, 2020; Song et al., 2019; Vanderzalm et al., 2016). These reactions are complex, however, and characterizing subsurface conditions sufficiently to predict them is costly. More information is needed on the range of geochemical reactions that can occur in the vadose zone and aquifers, as well as tools and frameworks that can be used to identify these risk factors in the field.

A promising approach to risk mitigation that speaks to the lack of any national trends is a local, risk-based site evaluation framework. For example, the state of California’s framework for the siting and design of dry wells (Geosyntec, 2020) characterizes sites as “high,” “medium,” or “low” risk based on the type of land uses in the watershed (e.g., industrial tends to be “high” risk), the presence of existing sources of contamination (e.g., septic tanks, USTs), and other hydrogeological factors that may lead to rapid transmission of contaminants to an aquifer. While limited to California and dry wells, the framework provides a template that could be used for advancing concepts of source water protection more broadly to other stormwater EAR practices.

What are locally appropriate, fit-for-purpose pretreatment options that reduce the risk of groundwater contamination?

Historically, water resources managers have placed less emphasis on treatment of stormwater (compared to untreated wastewater) in EAR systems. Historically, pretreatment has largely been incorporated to address factors that could lead to clogging of surficial soils or well screens, such as sediment and nutrients. This has established a reasonable knowledge base on the effectiveness of typical pretreatment systems, such as settling basins and constructed wetlands, at removing sediment and nutrients.

However, as discussed throughout this report, stormwater can carry with it a number of other pollutants that can be harmful to humans and the environment and that can persist from days to decades. Much less is known about the ability of typical pretreatment systems to effectively reduce or remove these pollutants, such as pathogens and trace organics. Identification of locally appropriate or fit-for-purpose pre-treatment technologies is needed to mitigate risks posed by specific contaminants. It is critical that knowledge of pretreatment effectiveness be combined with a better understanding of the presence and risks associated with stormwater pollutants in a variety of development and land use settings. The stormwater community can learn about the performance of more advanced treatment processes, such as media filtration, from the wastewater community, but more research is needed on stormwater-specific applications of these technologies under diverse development and hydrogeologic conditions. Such research should include site-specific work on how such technologies function under a range of recharge conditions and include consideration of existing as well as new pretreatment technologies.

How will EAR systems perform over time?

Finally, monitoring of stormwater EAR projects is needed to understand the current and long-term performance of EAR systems, including operation and maintenance needs. Long term monitoring is also important for evaluating the potential effects of changing environmental conditions, such as climate and land use change, on infrastructure. Stormwater, groundwater, and (for ASR operations) recovered water should be monitored. The system should be adequately characterized in terms of infiltration rates, underground transmission routes, and potential for chemical reactions between infiltrated stormwater and aquifer substrate. Modeling should also be applied where appropriate to support this need. Protection of groundwater resources would be furthered by programs to develop monitoring networks for critical control points for chemical and microbial contaminants. Given the potential for stormwater contamination to derive from spills (including sewage, fertilizers, pesticides, and industrial chemicals), real-time monitoring approaches that leverage wireless technology and novel sensors should be evaluated at critical control points to notify managers of potential impacts. Modeling can also help characterize risk of future impacts from changing climates. Most stormwater systems today are designed and constructed assuming steady state climate patterns for temperature, total rainfall, storm event frequency, and storm event intensity. Depending on the location, each of these metrics could change, which could have consequences for the function and effectiveness of stormwater EAR systems. These issues are discussed in Section 4.1.4, however more research is needed to ensure stormwater EAR planning and design strategies are effective under current and future climate conditions.

8 SUMMARY AND CONCLUSIONS

EAR of stormwater is potentially a cost-effective way of increasing the resilience of drinking water supplies. Stormwater EAR can also play a role in supporting the health of natural ecosystems, and in meeting other regional or local needs such as mitigating the effects of saltwater intrusion, helping to prevent subsidence due to excessive groundwater withdrawal in some areas, and reducing discharge to surface waters during storms.

Increasing populations, urban development, and climate change are putting more pressure on water resources, meaning that the value of stormwater EAR may increase in the future. Capturing and storing untreated stormwater is critically different from capturing and storing treated wastewater or drinking water, in that stormwater can contain a significant number of chemical and microbial contaminants that could be detrimental to receiving aquifers (e.g., Masoner et al., 2019). Accordingly, EAR of stormwater carries potential risk. This report is a review and synthesis of scientific and technical literature on EAR using stormwater. Our goal is an improved understanding of the scientific foundation, including knowledge gaps, leading to best practices for effective and safe EAR using stormwater under diverse development and hydrogeologic conditions.

Available literature shows EAR projects vary in size, design, and performance. EAR has been implemented successfully, but results are highly site-specific, depending on local hydroclimatic and geologic setting, project design, maintenance requirements, and operating costs. EAR brings significant benefits to communities regardless of the original intent of the EAR project. For example, “mining” of aquifers in Arizona has left some communities struggling to ensure a sustainable water supply and has degraded the water quality of remaining groundwater. Dry wells used for stormwater EAR in Arizona have been shown to increase recharge rates more than tenfold compared to predevelopment conditions. This successful recharge of aquifers helps to counter groundwater extraction associated with development pressure (Milczarek et al., 2005). Similar benefits are realized in Stockton, California, where infiltration basins are used to recharge public supply wells. These basins have provided dual benefits: flood control and stormwater capture during the wet season and diversion and recharge of nearby surface water during the dry season (O’Leary et al., 2012).

Examples of EAR systems using stormwater that reduce flood risks abound as well. In Nassau County, New York, infiltration basins have increased the levels of the local water table by 5 feet compared to predevelopment levels (Bouwer et al., 2008). In New Mexico, a repurposed mine site provides flood risk reduction and recharges 30–50% of onsite precipitation (Miller, 2006). In another study, urban low-impact development was implemented to reduce urban stormwater effects on downstream waters, reduce flooding, and replenish groundwater supplies. The project treats roughly 220 acres of contributing area, provides about 41 acre-feet per year of groundwater recharge, and results in a 96% reduction in pollutant discharge (due to runoff reduction) that is credited towards the local total maximum daily load (Sadeghi et al., 2018).

To ensure that groundwater resources are protected, and given the significant difficulties associated with remediating aquifers that do become contaminated, EAR projects using stormwater must consider the larger watershed/aquifer context. While the past few decades have seen significant progress in the remediation of contaminated rivers, lakes, and other surface waters, the subsurface presents access challenges. Preventing groundwater contamination is significantly easier than remediating groundwater.

Pathogens, including bacteria, protozoa, and viruses, constitute one group of contaminants that pose a potential risk to human health and the environment. Because individual pathogenic organisms are difficult to measure, FIB are used to screen for the presence of possible pathogenic organisms. Generally, stormwater from urban and high-density areas tends to have higher concentrations of FIB (Ahmed et al., 2019). Once in surface waters or aquifers, viruses and protozoa tend to persist longer than bacteria (Pitt et al., 1999; Sidhu et al., 2015), though longer residence times tend to promote attenuation of all groups through physical retention, inactivation, and dilution (Maliva, 2020; Zhang et al., 2013). The pathogens of greatest concern in groundwater are enteroviruses, *Shigella*, *Pseudomonas aeruginosa*, and various protozoa, such as *Cryptosporidium* (Clark and Pitt, 2007; Pitt et al., 1999, 2003).

Organics and metals are other groups of compounds that are commonly found in stormwater. Common organics include pesticides, the herbicide diuron, and the PAHs fluoranthene and pyrene (Masoner et al., 2019; Pitt et al., 1995). Common metals include lead, zinc, copper, and cadmium (Pitt et al., 1994). As stormwater is often associated with urban impervious surfaces such as roadways, driveways, and parking lots, metals and PAHs that derive from vehicle operation and wear, as well as asphalt deterioration and parking lot sealcoat (Mahler et al., 2005), are very common in stormwater. A nearly 40-year-old stormwater study found that the main organics in stormwater included fluoranthene, pyrene, phenanthrene, bis-(2-ethylhexyl) phthalate, and pentachlorophenol (U.S. EPA, 1983). These contaminants continue to be found in more recent studies. PFAS constitute another group of organics more recently considered as part of stormwater quality studies. While less is known about these compounds, they may pose a greater risk over the longer term given their extreme persistence. There is little to no sampling of aquifers adjacent to infiltration sites and potential impacts to groundwater quality need to be studied.

A number of options exist to mitigate the risk posed by the range of contaminants potentially found in stormwater. Traditional stormwater treatment practices (such as settling basins, constructed wetlands, and green infrastructure) all have demonstrated abilities to remove pollutants like nutrients, metals, and PAHs, and are commonly used for stormwater EAR pretreatment. Their ability to remove pathogens or more complex and persistent contaminants like PFAS compounds is less known, and adoption of more advanced pretreatment systems like media filtration and chemical additions may be necessary if stormwater EAR becomes more widely adopted. Local risk-based frameworks that can be used to screen for hazardous land uses, existing sources of contamination like septic tanks and USTs, and problematic hydrogeological features can also help mitigate risk of aquifer contamination.

Stormwater EAR has tremendous potential to address a number of water resource problems, from improving the resiliency of public water supplies to restoring groundwater flows to degraded ecosystems, all while providing flood protection and water quality improvement. However, stormwater EAR in the United States is still in its infancy, and more widespread adoption carries a number of significant risks. This report provides a benchmark for the current understanding of the scientific and technical information now available to practitioners. It also highlights current knowledge gaps that, if filled, will help inform recommendations for effective and safe stormwater EAR.

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