FEASIBILITY STUDY AND NUMERICAL MODELING ON MANAGED AQUIFER RECHARGE USING DRY WELLS IN HARRIS COUNTY, TEXAS

A Thesis

by

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Submitted to the Office of Graduate and Professional Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

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August 2018

Major Subject: Civil Engineering

ABSTRACT

Managed aquifer recharge (MAR) has been adopted as a modern water management strategy to store water into subsurface during surplus periods for future recovery. As one application of MAR, dry well is installed in vadose zone and can be used to facilitate infiltration of recharge water through an unsaturated formation into the underlying aquifer. In this study, we investigated the feasibility of applying dry wells in Harris County, Texas. Harris County is located in southeastern Texas and contains Houston, the fourth largest metropolitan area in the US. The county suffered both severe floods and droughts in recent years, including Hurricane Harvey in 2017, and drought in record in 2011.

This project was performed in two phases. Phase 1 consists of feasibility study, in which the suitable recharge locations were investigated based on existing data. Phase 2 focuses on modeling the dry well performance using HYDRUS 2D, for a range of soil conditions and physical configurations, to assist in developing guidelines for dry well design and its application in Harris County.

In feasibility study, central portions of Harris County were identified to be suitable for dry wells. Following low impact development (LID) concept, which aims to reduce runoff near the source, dry wells are recommended to be placed near existing retention basins. In model simulations, three saturated hydraulic conductivities (K_{sat}) of soils from targeted sites were incorporated (0.24 m/d, 0.47 m/d, and 0.78 m/d). Aiming for an extensive investigation on the controlling factors, the well length, duration of injection, separation distance between dry well and water table, multiple injections, antecedent soil moisture, K_{sat}, natural recharge, location of clay lens, and tracer movement, were addressed in modeling process. The model suggests dry

wells are most appropriate for regions with a deep groundwater table, if the high recharge rate is of the uppermost concern. High antecedent soil moisture can result in slow water movement and limited storage capacity. In certain areas, injecting several days after the peak storm is preferred if timed operation is possible. Prior to dry well construction, location and size of subsurface clay lens should be identified to avoid negative impact from clay lens. Contaminant fate and transport tests are recommended to determine the minimum separation distance for various ranges of soil $K_{sat.}$

ACKNOWLEDGEMENTS

I would like to give special thanks to my committee chair, Dr. Gretchen Miller, for always being supportive, guiding me patiently through the study, sharing insightful ideas, and encouraging me with her enthusiasm for research. And thanks to my committee members, Dr. Olivera and Dr. Sheng, for their guidance and support throughout the course of this research.

Thanks also go to my research group members, Saheli Majumdar, Tayyab Mehmood, Courtney Corso, Cody Saville, Ben Smith, and Jaeyoung Song, for sharing so much valuable advice, and offering support whenever needed. I would extend my gratitude to my friends and roommates, who made my time at Texas A&M University a wonderful experience.

Finally, thanks to my dear Mom and Dad, without their support I would not have been able to follow my passion and go this far. And to my dog, Mars, who has been my cutest companion since I started this project, and who is also part of this work with his name (Managed Aquifer Recharge using Stormwater).

CONTRIBUTORS AND FUNDING SOURCES

Contributors

This work was supervised by a thesis committee consisting of Professors Gretchen R. Miller and Francisco Olivera of the Department of Civil Engineering and Professor Zhuping Sheng of the Department of Biological and Agricultural Engineering.

All work for the thesis was completed independently by the student. The data collection process was guided by Professor Gretchen R. Miller, and the numerical modeling was under the advisement of Professors Gretchen R. Miller and Zhuping Sheng.

Funding Sources

This work was made possible in part by Harris County Flood Control District via subcontract with Binkely and Barfield, Inc.

Its contents are solely the responsibility of the authors and do not necessarily represent the official views of the Harris County Flood Control District.

NOMENCLATURE

ADEQ	Arizona Department of Environmental Quality
BMPs	Best Management Practices
CEC	Cation Exchange Capacity
DEM	Digital Elevation Model
EC	Electrical Conductivity
EPA	U.S. Environmental Protection Agency
GCA	Gulf Coast Aquifer
HCFCD	Harris County Flood Control District
HCFWS	Harris County Flood Warning System
HGSD	Harris-Galveston Subsidence District
LID	Low Impact Development
MAR	Managed Aquifer Recharge
MCLs	Maximum Contaminant Levels
NJDEP	New Jersey Department of Environmental Protection
OEHHA	Office of Environmental Health Hazard Assessment
TCEQ	Texas Commission on Environmental Quality
TWDB	Texas Water Development Board
TWRI	Texas Water Resources Institute
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
VOCs	Volatile Organic Compounds

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CHAPTER I

INTRODUCTION

1.1 Background

In recent years, with rising water demand around the world, and erratic climate pattern which has caused frequent flood and drought, storing water during the surplus periods to help relieve water scarcity in dry days has become a modern water management strategy. Managed aquifer recharge (MAR) using stormwater is one application for such purpose. The application of MAR is to infiltrate source water into the soil and let water move downward to recharge aquifer or to directly inject water to aquifer through wells (Bouwer 2002). Compared to storage via dams, MAR is able to reduce evapotranspiration losses to near zero, halt land subsidence and seawater intrusion, support groundwater-dependent ecosystems, and improve groundwater quality through porous media filtration (Bouwer 2002, Dillion 2009).

In this study, we investigated the feasibility of MAR using excessive stormwater in Harris County, Texas, focusing on the use of dry wells as an engineering solution. Recently, Harris County suffered both severe floods and droughts. The most recent Hurricane Harvey has set several records in the nation. The devastating hurricane started to hit Houston area on August 25th 2017 and caused three massive rainfalls in six days. On Sept. 1st when peak rainfall occurred, one-third of Houston was under water (Amadeo 2017). The Cedar Bayou, located in eastern Houston, received a total rainfall of 51.88 inches, setting a record for a single storm in continental U.S. (Amadeo 2017). In 2001, the tropical storm Allison dumped as much as 80 percent of annual average rainfall in Harris County, and caused over $5 billion in property damage (HCFCD 2001). The county experienced the driest year in record in 2011. From October

2010 to September 2011, City of Katy (western Harris County) received a 12-month precipitation of 14.67 inch (Lindner 2012). With the uncertainty of future climate, application of MAR would prepare the county for potential droughts while appropriately managing excessive water during storm events.

1.2 Objectives

The main goals of this project are to identify appropriate sites for dry well practice in Harris County; and to model the performance of dry well on selected sites, focusing on the quantity of recharged water. Main tasks include collecting data from publicly accessible sources; mapping in ArcGIS (e.g. soil properties, depth to groundwater table, etc.); modeling single dry well performance under various conditions (e.g. soil moisture, well dimension, schedule of injection, etc.) in HYDRUS 2D. The findings from this study can be used for planning future pilot studies and assisting in the establishment of local dry wells standards.

1.3 Literature Review

Long term groundwater depletion has caused severe regional scarcity of groundwater and land subsidence. MAR has been applied to address such concerns. In Central Valley, California, total depletion of groundwater during the last century was estimated as 154 km^3 (Scanlon 2016). In some parts of the valley, land subsidence up to 10 m has resulted in less storage capacity in subsurface (Ireland 1986; Sneed et al. 2013). In south central Arizona, groundwater depletion created groundwater level declines up to 30 to 120 m in different regions, and land subsidence up to 6 m was observed (Anderson et al. 1992; Galloway et al. 1999). In both Central Valley and Arizona, application of MAR has reversed groundwater level decline trends (Scanlon 2016).

Dry wells are an infiltration practice located in vadose-zone and can be used to facilitate infiltration in unsaturated formation. It is a borehole or circular pit that is cased and screened above the water table of an unconfined aquifer. During injection, water may be purified while moving through the buffer zone between dry well and water table, and ultimately reaches the groundwater. Compared to other common MAR techniques, dry wells are more economically efficient than deep injection wells, and have less land requirement than infiltration basin systems (Bouwer 2002).

Dry well is classified as class V injection well by EPA, which is conduit placing non-hazardous fluid directly to subsurface (Pi et al. 2017). Class V injection wells can be used to dispose water, or inject high quality water into an aquifer for storage and subsequent recover. EPA has no design requirements on dry wells, and dry wells can be operated as long as they are registered with EPA and the operation poses no threat to underground sources of drinking water (Pi et al. 2017). In Texas, the EPA and the Texas Commission on Environmental Quality (TCEQ), and rules of the Harris-Galveston Subsidence District (HGSD 2016) provide regulatory parameters for MAR programs in Harris County. TCEQ rules require a permit for new injection wells or converting existing wells into an injection well (TCEQ 2018). To implement such MAR project, acquiring a TCEQ water right permit and complying with EPA and TCEQ Class V Injection rules are required.

Since pollutants carried by storm runoff can directly enter dry wells without purification by surface or near surface soil, there is concern of deterioration of local groundwater. In the EPA's 1999 Class V wells report, concentrations of heavy metals and organics in stormwater exceeded regulatory levels (EPA 1999). Although high concentrations of contaminants were detected in storm runoff, groundwater samples near dry well sites did not show similar levels (Edwards et al. 2016). Wilson et al. (1990) performed an investigation on potential pollution of dry wells in Tucson, Arizona, and they found there is no evidence showing volatile organic compounds (VOCs) accumulate in the vadose zone at study sites. In Modesto, CA, over 11,000 dry wells have been constructed since 1950s to distribute stormwater in an agricultural and urban setting. The groundwater quality studies in Modesto found no contaminants detected exceeded the regulatory levels, although chemistry of water has been affected by agricultural activities (Jurgens et al. 2008). The City of Park Ridge, Wisconsin, distributes the stormwater through dry wells without pretreatment design (Lindemann 1999). In the water quality study near dry well sites, concentrations of polycyclic aromatic hydrocarbons (PAHs) were consistently high, and the highest PAHs was detected in the runoff in dry well closest to a major highway (Lindemann 1999). The results showed PAHs were absorbed by sediment at the dry well bottom, and it was recommended that sediment at the well bottom be sampled and disposed every few years (Lindemann 1999). Edwards et al. (2016) provides an extensive water quality literature review regarding to dry wells practices.

In urban area, the most common pollutants carried by stormwater are oils and grease (form vehicles), bacteria such as *E.coli* and fecal coliforms, nutrients (P, NH_4, TKN), and heavy metals (Pb, Hg, Cr, As, Cu) (Pitt et al. 2005). Metals were found to be more commonly above advisory levels, and urban “hot spots” containing higher concentrations of trace metals include gas stations, industrial sites, and convenience store parking areas (Edwards et al. 2016, Pitt et al. 1996). In stormwater infiltration practices, Pitt et al. (1999) identified pollutants with high mobility in subsurface, and high soluble fractions, such as nitrate, flouranthene, pyrene, and chloride, should be taken into great account. However, in a nationwide storm runoff study, EPA concluded that runoff contaminants variability can be more consistently explained by the intensity and duration of storm than location and land use variability (EPA 1983).

Although studies showed stormwater contains various kinds of contaminants, most of the studies concluded that with sufficient natural pollutant attenuation and enough separation distance, application of dry wells with stormwater poses no threat to groundwater quality (Edwards et al. 2016). In dry wells practices, water quality monitoring and specific design standards are often implemented. Oregon requires a 500 feet setback or 2-year travel-time between dry wells and water supply wells, and a 5 feet separation distance between dry well and groundwater table (OEHHA 2018). In California, dry well is recommended to be at least 100 feet away from public supply wells, and separation distance should be at least 10 feet (Pi et al. 2017). New Jersey specifies a permeability rate between 1 ft/day and 20 ft/day, and a separation distance of 2 feet (NJDEP 2016). However, in a water quality study in Millburn Township, New Jersey, there was little to no difference in water quality between samples collected from well bottom and from 2 feet below well bottom, indicating 2 feet of separation distance had negligible impact on contaminants attenuation (Pitt and Talebi 2012).

Studies have shown the risk of polluting groundwater and drinking water can be minimized if dry wells are used at suitable places with proper maintenance (Edwards et al. 2016). Pretreatment is often included in dry well design to reduce sediment load, prevent clogging in the well, and extend the functional life of the system. Vegetated buffer followed by a sediment tank is commonly applied. EPA (1999) evaluated engineered pretreatment techniques, including filter strips, vegetated swales, vegetative infiltration basins, infiltration trenches, sand/gravel filters, wet ponds, stormwater wetlands, and porous pavements. Studies showed porous pavements have the highest removal rates, followed by infiltration trenches, sand filters, grassy swales, and detention basins (EPA 1999). It was found pretreatment generally is more effective in removing sediments, lead, copper, and zinc, than water soluble pollutants such as nitrogen, nitrate, and phosphorus (EPA 1999). In pretreatment structure, maintenance access is required by New Jersey stormwater BMP to reduce sediment and vegetation loads entering the dry well (NJDEP 2016). In Elk Grove, CA, 50-65% of total suspended solids was removed by grassy swale or water quality basin near the dry wells (Nelson et al. 2017). One study performed evaluations of effectiveness of different stormwater best management practices (BMPs) in Maryland. Of 258 BMPs, 22 were dry well sites. Other BMPs in competition were infiltration basins, wet detention basins, vegetated swales, etc (Lindsey et al. 1992). Dry wells were proved to have higher success than others in multiple categories: most of the dry wells (20) had water quality benefits; least sites (6) needed maintenance; none of dry wells had structure failure; and least sediment entered facilities. (Lindsey et al. 1992).

Typical dry well construction includes 1 m diameter perforated concrete or PVC pipe, depth from 0.6 to 26 m, minimum 1.5-3 m separation distance from well bottom to seasonal high water table to ensure enough residence time (Figure 1) (Bouwer et al. 2008; City of Portland 2014; Jurgens et al. 2008; Lindemann 1999; OEHHA 2018; Pitt et al. 2012; Wilson et al. 1990). In California, soil should contain less than 30% clay or less than 40% silt; dry wells should be set a minimum of 100 feet apart; and the site slope should be less than 15% (Pi et al. 2017). According to New Jersey stormwater BMP manual, when planning a dry well, soil characteristics, separation distance, sensitivity of the region, and inflow water quality are the key factors to be considered (NJDEP 2016). In years of study at a site in California, Harter et al. (2005) concluded that stratified layers in vadose-zone has the advantage in slowing contaminant transport. The effectiveness depends on hydrogeology and land use near the site, and quantity and quality of injected water (Edwards et al. 2016). In some regions, preexisting dry wells standards have been reformed based on dry well practice (Brody-Heine et al., 2011; City of Portland 2014; Wilson et al. 1992).

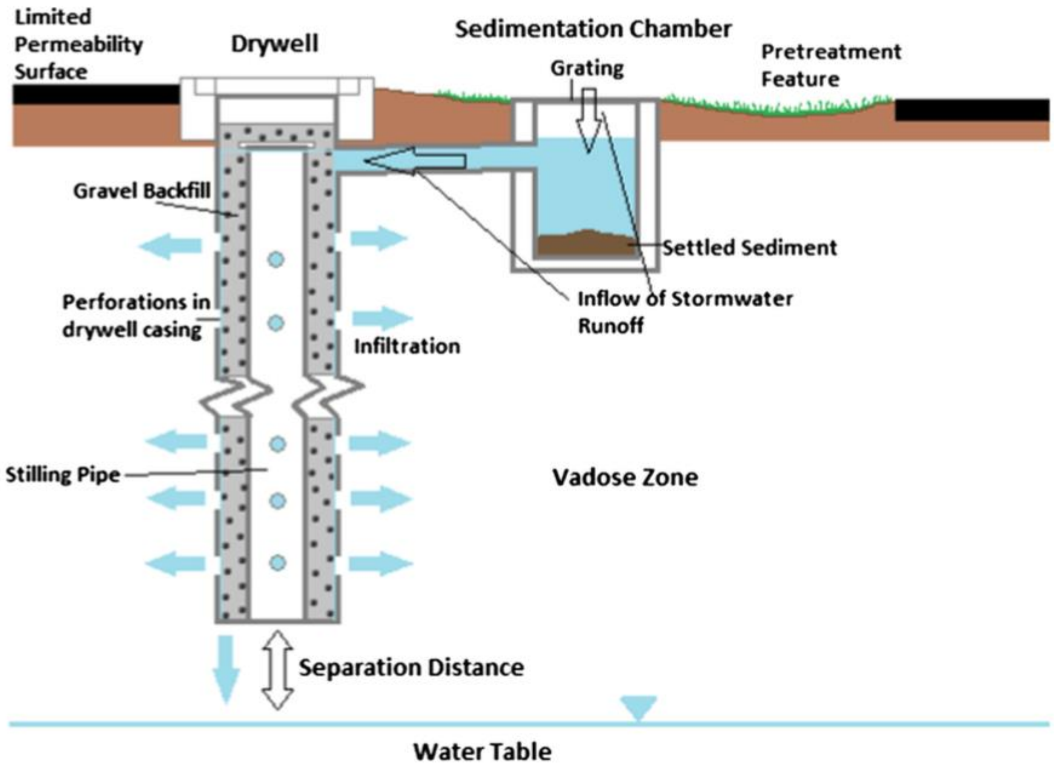

Figure 1. Typical design of dry well system (Reprinted from Edwards et al. 2016)

The existing MAR systems in Texas mainly apply infiltration basins and deep injection wells for recharge. In the U.S., dry wells are increasingly being adopted as a stormwater management tool, most notably in Oregon, Arizona, and California. Selected examples are as follows.

Portland, Oregon. Dry wells have been applied extensively as the major tools for managing stormwater. Approximately 9,000 dry wells are spread throughout the city (OEHHA 2018). Snyder et al (1994) estimated 75% of precipitation falling on impervious areas was collected by dry wells, which is 38% of total groundwater recharge in the city. Stormwater is collected in a catch basin (street gutter, vegetated buffer, etc.), filtered through a sedimentation manhole of 3-4 feet in diameter and 10 feet deep, and released into a dry well of 20-40 feet long (Figure 2). Oregon is known for having developed dry wells programs with effort to protect groundwater quality. Placement of dry wells should avoid vehicle maintenance areas and gas and fire stations, and 500 feet away from water supply wells (OEHHA 2018). Drinking water standards such as maximum contaminant levels (MCLs) are applied to control the quality of inflow to dry wells (OEHHA 2018). The dry wells are operated by municipalities. The regulatory program consists of minimum two years of monitoring to ensure concentration in runoff are within criteria, conservative one-dimensional groundwater fate and transport modeling, and annual report describing monitoring results (OEHHA 2018).

Figure 2. Typical city dry wells system in Portland (Reprinted from OEHHA 2018)

Phoenix, Arizona. Dry wells first appeared in the city in 1930s as a stormwater control strategy. By end of 2014, Arizona Department of Environmental Quality (ADEQ) had 51, 507 registered dry wells, and 95 percent of which locate in the greater Phoenix area (Graf 2015). ADEQ requires at least 10 feet of separation distance. In City of Chandler, southeast of Phoenix, 3,763 dry wells were installed within city boundaries to drain stormwater from 1,400 acres of retention basin. The estimated recharge through dry wells was 2,100-3,100 ac-ft/year (Graf 2015).

Scottsdale, Arizona. It took the City of Scottsdale nearly 5 years to perform tests and studies prior developing the full scale of recharge system. During the test, it was suggested that the system could allow hundreds of dry wells distributed 50 feet apart (Bouwer et al. 2008). In the final design, dry wells were drilled to a depth of 180 feet with depth to groundwater of approximately 600 feet (Bouwer et al. 2008). To prepare for various source water quality, two types of dry wells were constructed. The wells using polished water are cased with PVC, while the emergency wells receiving water with higher particulate matter are equipped with socks filled with gravel pack (Bouwer et al. 2008). In flow test, the primary wells achieved average recharge rates of 450 gpm, and the emergency wells provided average rate of 300 gpm (Bouwer et al. 2008). Currently, the city recharges the treated wastewater through a well field with 63 injection wells, and lets the water flow 400 feet through vadose zone before reaching the drinking water aquifer (City of Scottsdale 2018). Scottsdale recharges over 1.7 billion gallons of recycled water to augment future drinking water supplies (City of Scottsdale 2018).

Modesto, California. Dry wells have been applied as a primary stormwater management strategy to distribute stormwater in agricultural and urban areas since 1950s (Jurgens et al. 2008).

Over 11,000 dry wells have been constructed in the incorporated area in City of Modesto. The dry well design incorporates 1 m diameter of drilled holes with 15-25 m deep, and a catch basin adjacent to each well to capture storm runoff and deliver water to well when overflowed (Jurgens et al. 2008).

Elk Grove, California. In a pilot study performed by Nelson et al. (2017) in Elk Grove, two dry wells were constructed in an interbedded sand and clay at a water quality basin, with depth to water table of 48 and 74 feet. The estimated mean volume of water passing through a dry well was 0.7 acre-feet, based on yearly rainfall of 13.5 inches (Nelson et al. 2017). Depending upon storm size, initial infiltration rates varied from 20 to 97 gpm with 1,100 to 28,400 total gallons recharged per event, and these decreased as rainy season progressed and soils became saturated (Nelson et al. 2017).

Millburn Township, New Jersey. The town is close to New York City, and has average annual rainfall of 44 inches (Pitt and Talebi 2012). Millburn has required to apply dry wells for flood mitigation since 1999, and there are over 1500 residential and commercial dry wells (Pitt and Talebi 2012). The dry wells are 6 feet in diameter and 6 feet deep with side openings and open bottom, rest on a 2 feet layer of crushed stone, and have 2 feet of separation distance (Pitt and Talebi 2012). Many of the dry wells are without cover, and are located in landscaped areas to receive runoff from roof drain leaders and storm drains in driveways or small parking lots (Pitt and Talebi 2012). The study on water quality showed 2 feet of separation distance created minimal attenuation on contaminants, and pretreatment design was recommended (Pitt and Talebi 2012).

1.4 Study Overview

This project was performed in two phases. Phase 1 consists of feasibility study, in which the suitable recharge locations was investigated based on existing data (see Chapter 2). Phase 2 focuses on modeling the performance of dry well using HYDRUS 2D (see Chapter 3).

In Phase 1, following data were compiled from various sources:

- Soil properties: hydraulic conductivity, electrical conductivity, cation exchange capacity (USDA 2017).
- Locations and usage of wells in Chicot aquifer (TWDB 2016).
- Depth to water table, change in piezometric surface, and hydraulic gradients in Chicot aquifer (Kasmarek et al. 2015).
- Specific yield, storativity, and transmissivity in Chicot aquifer (Kasmarek et al 2012).
- Measured land subsidence during 1915-2001 (Kasmarek et al. 2012)
- Watershed boundaries, floodplains, existing detention basins (HCFCD-BMP 2017)
- Water quality: impaired waterways, toxic releases to land, toxic releases to water (EPA).

Following low impact development (LID) practices, which requires removing storm runoff at its source and minimize interference on ecosystem, primary sites are near existing detention basins. Based on preliminary assessment in Phase 1, central and south-central Harris County appear to be suitable for dry well application. Specifically, three detention basins in central Harris County, E500-01-00, E500-06-00, and E500-10-00, were selected for extensive study. The averaged soil saturated hydraulic conductivities in these basins were used in HYDRUS 2D simulation.

In Phase 2, the HYDRUS 2D model was applied to assess dry well performance under various conditions. The factors evaluated include well length, duration of injection, separation distance, multiple injections, antecedent soil moisture, saturated hydraulic conductivity, natural recharge, and tracer movement. HYDRUS is a finite element code using Richards Equation and Fickian-based advection-dispersion equation to model water and heat movement in vadose zone (Simunek et al. 2012). Studies have shown HYDRUS is capable of simulating water and heat movement in vadose zone, and saturated aquifer system. HYDRUS is a premier modeling choice for this study since it was designed specifically for simulation of infiltration and recharge processes in the vadose zone. The hydraulic method used was van Genuchten-Mualem.

CHAPTER II

HOUSTON HYDROGEOLOGY

Harris County is located above the northern Texas Gulf Coast Aquifer (GCA) (Figure 3), part of the regional Gulf Coast Aquifer System. The GCA is composed of three sub-aquifers (from youngest to oldest): the Chicot, the Evangeline, and the Jasper (Figure 4). The aquifers are composed of laterally discontinuous deposits of gravel, sand, silt, and clay (Kasmarek et al. 2015). Sediments in each aquifer are: Holocene- and Pleistocene-age sediments in Chicot Aquifer; Pliocene- and Miocene-age sediments in Evangeline Aquifer; and Miocene-age sediments in Jasper Aquifer (Baker, 1979, 1986). The sediments were deposited by fluvial-deltaic process and relocated by fluctuations of sea level (Kasmarek et al. 2015). With no confining layer between Chicot aquifer and Evangeline aquifer, the aquifers are hydraulically connected. However, groundwater flow between Evangeline and Jasper aquifer is restricted due to the Burkeville confining unit. Geologically, a significant number of growth faults parallel the Gulf Coast affected sediment accumulation and dispersal. Salt domes are also more common in north part than south part of the aquifer (Mace et al. 2006).

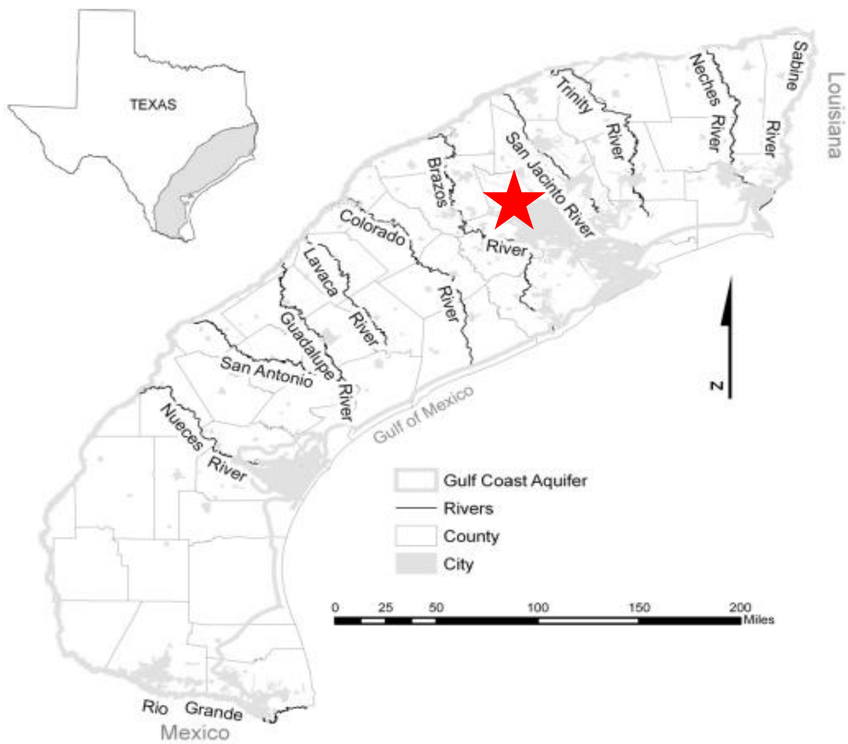

Figure 3. Texas Gulf Coast Aquifer, with star indicating the location of Harris County, (Reprinted from Kasmarek 2012)

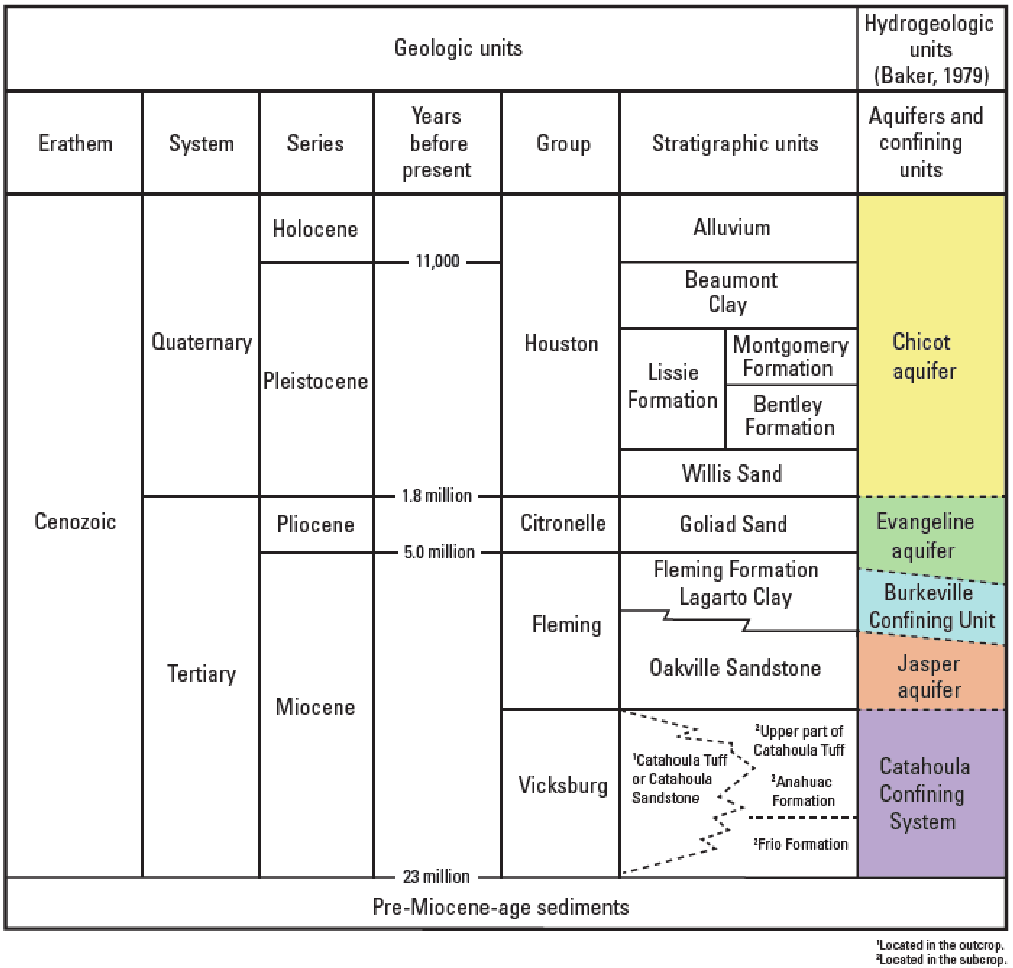

Figure 4. Geologic and hydrogeologic units of the GCA system in the Houston-Galveston region (Reprinted from Kasmarek et al. 2015)

In Harris County, water recharged via dry wells will be stored in the Chicot Aquifer. The Chicot Aquifer is made up of surficial alluvium deposits (from youngest to oldest): Beaumont Formation, Montgomery Formation, Bentley Formation, and Willis Formation (Kasmarek 2012). The estimated transmissivity ranges from 3,000 to 50,000 ft^2/day, and storativity ranges from 0.0004 to 0.1 (Carr et al. 1985).

2.1 Soil Properties

Soil plays a critical role in allowing downward infiltration during injection, as well as retaining the water for future recovery. Areas with coarse soil texture are favorable since coarse soil enables fast downward movement of injected water. However, the texture should be fine enough to hold water at the recharge site for a long period as a storage tank. To assess feasibility following soil properties, including soil saturated hydraulic conductivity, electrical conductivity, and cation exchange capacity were delineated in Harris County region.

Surficial soil data was obtained from USDA web soil survey (USDA 2017), which is the weighted average based on all layers. The estimates are based on the soil characteristics observed in field, such as soil structure and porosity (USDA 2017). In the maps below, white spots represent water body or no available data.

In the saturated hydraulic conductivity map (Figure 5), low values of 0-0.2 m/d mainly locate at southern Harris County, representing clay and loams formations in surficial soil. Soil in the middle to northern parts has hydraulic conductivity of 0.2-1 m/d, which indicates a mix of clays and sands in soil. Based on classification of soils (Bouwer 1999), soils in middle to northern parts are typically sandy loams, loamy sands, fine sands. High values of 1-7 m/d appear on the north edge of the county, classified as sands and coarse sands.

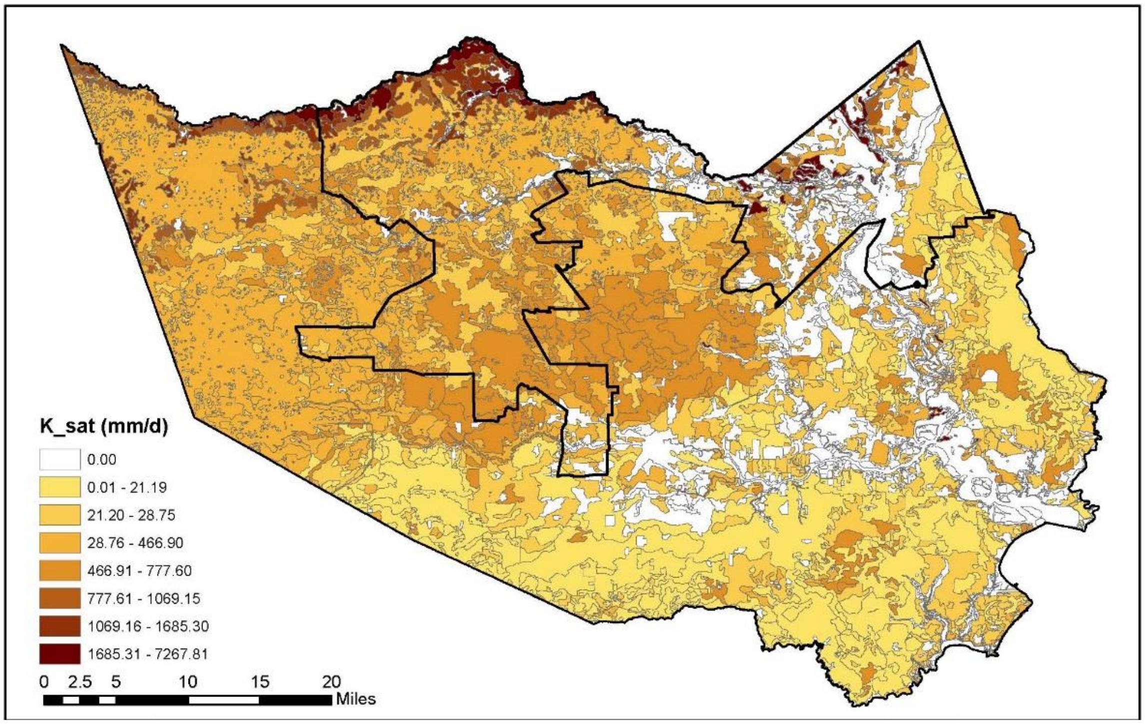

Figure 5. Soil saturated hydraulic conductivity, derived from USDA Web Soil Survey (USDA 2017)

Soil electrical conductivity (EC) is a measurement of concentration of water-soluble salts (e.g. sodium chlorine, sodium sulfate) in soil, and it is used to show saline soils (USDA 2017). Higher EC value indicates a higher salt concentration in soil water. Soil water salinity can cause fine particles to bind into aggregates, and may reduce hydraulic conductivity (Warrence et al. 2003). To ensure a high recharge rate during MAR application, lower EC sites are preferred. In the EC map (Figure 6), middle part has the highest EC, and generally northern Harris County has lower soil EC than the southern part.

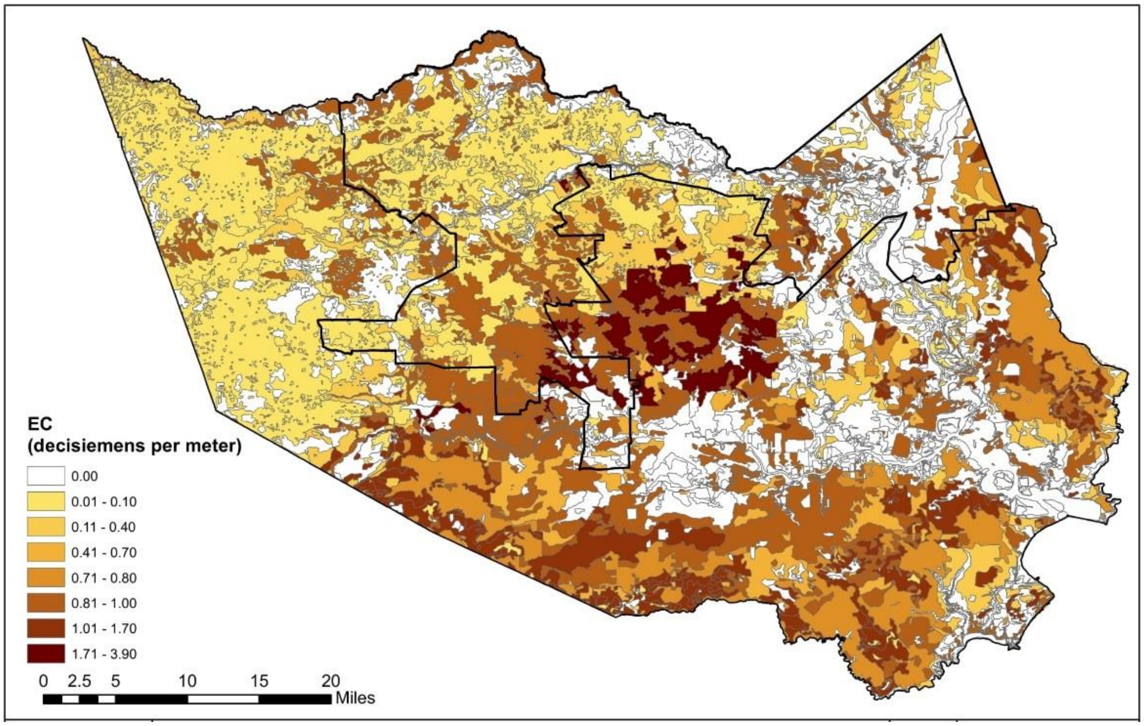

Figure 6. Soil electrical conductivity, derived from USDA Web Soil Survey (USDA 2017)

Another critical soil parameter is soil cation exchange capacity (CEC). CEC is defined as the total amount of extractable cations can be held by soil (USDA 2017). A higher CEC is favorable for reducing hazard of groundwater pollution (USDA 2017). The southeastern parts of Harris County have higher CEC values (Figure 7). In our interested area, the Precinct 4 (highlighted in black outline), southern parts of precinct 4 appear to have a better cation holding capacity.

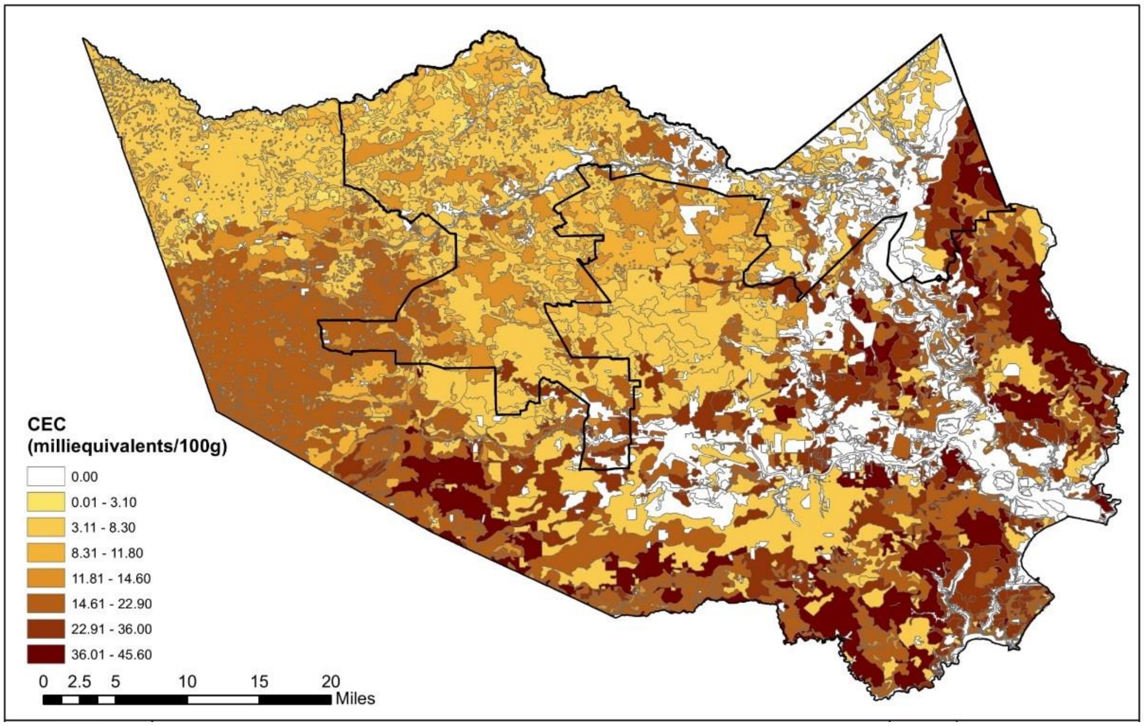

Figure 7. Soil cation exchange capacity, derived from USDA Web Soil Survey (USDA 2017)

2.2 Aquifer Properties

Due to the potential for water quality deterioration from dry wells, California requires that dry wells should be located with sufficient distance from existing wells (Pi et al. 2017). The shapefiles of location and use of existing wells are available on TWDB. In Chicot aquifer, wells for various uses distribute across the whole county in a highly dense format. For a clear view from map, the use of wells was specified into municipal well, irrigation well, industrial well, and domestic well. Locations of municipal wells into Chicot aquifer are shown below as an example (Figure 8). For future studies such as selecting specific sites for pilot studies, and predicting the impact of injection and recover on other well owners, data from existing wells has to be taken into account.

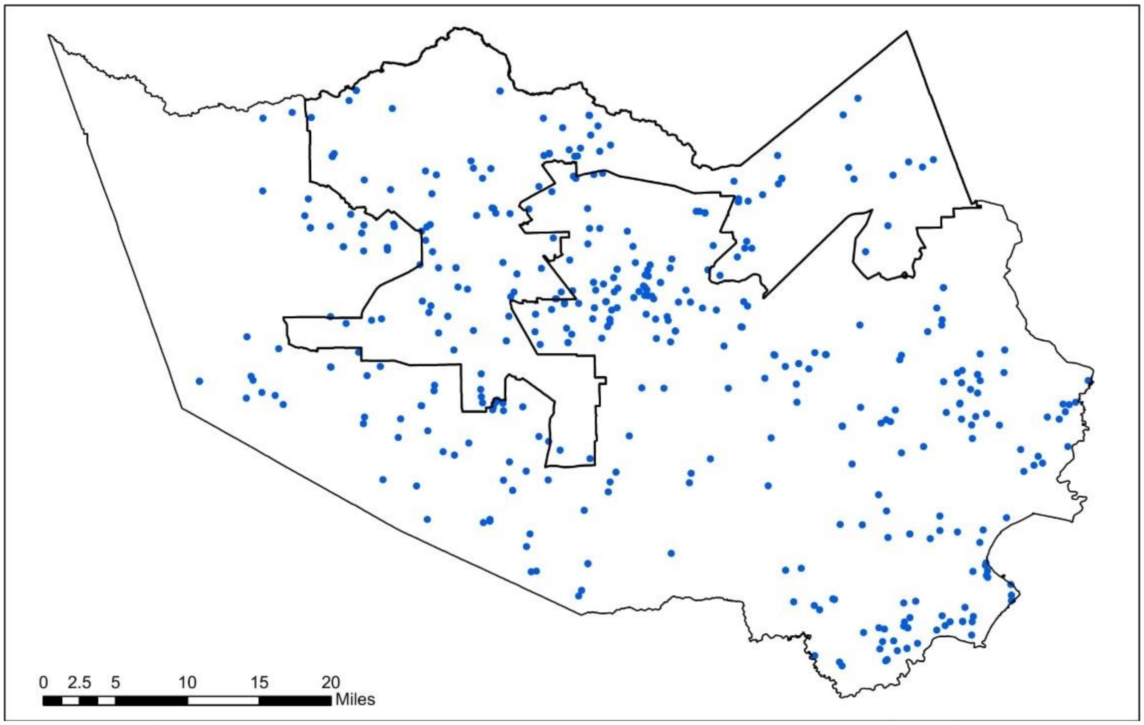

Figure 8. Location of municipal wells in Chicot Aquifer, derived from Texas Water Development board (TWDB 2016)

Depth to water table reflects the potential storage capacity, as well as assisting in evaluating the groundwater mounding after injection. A map showing the depth to groundwater (Figure 9) was created from piezometric surface data and digital elevation model (DEM). The inverse distance weighting method was used to generate the raster map, and the dots in map illustrate the data points used for interpolation. Piezometric surface levels were obtained from USGS, DEM was from USGS national map. Generally, central portions of the county have lower groundwater levels, because of extensive withdrawals in Houston during 1970s. The decline of groundwater in central county not only potentially enables sufficient storage capacity, but also provides the opportunity of controlling land subsidence through MAR.

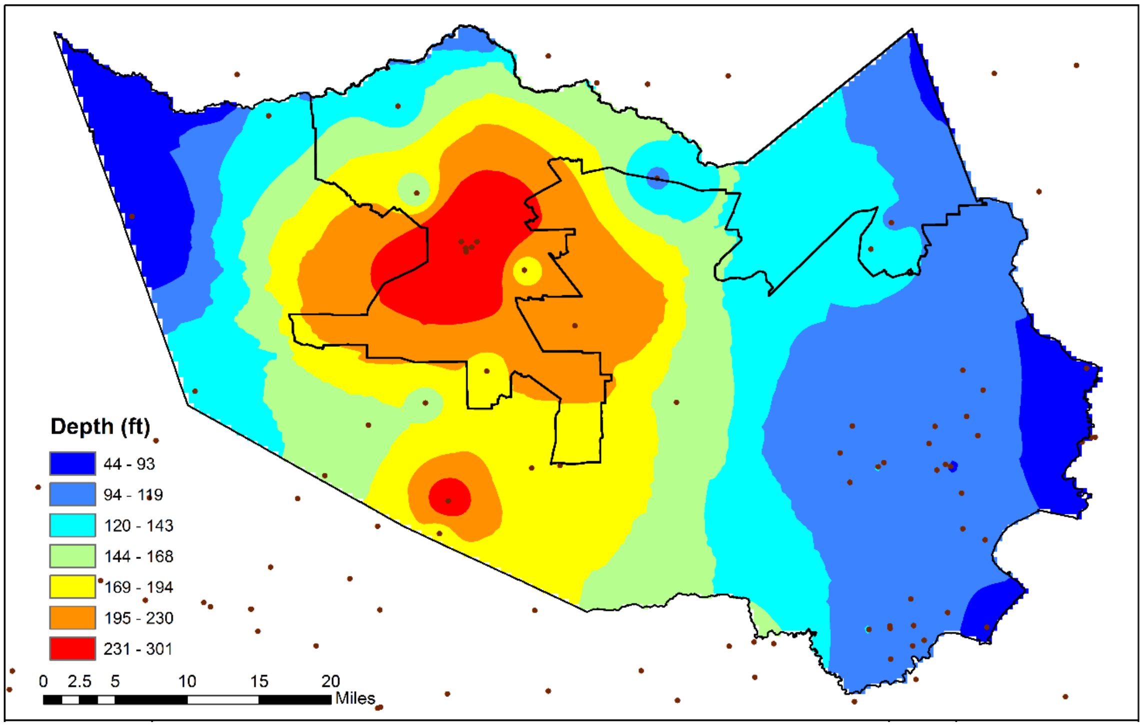

Figure 9. Depth to water table in Chicot Aquifer (Kasmarek et al. 2015)

From the piezometric surface map, hydraulic gradients (Figure 10) were delineated to show groundwater flow direction. Western to northwestern portions of the county have higher hydraulic gradients. The southern part of precinct 4, as suggested as suitable for MAR in soil analysis, appears to have stable groundwater level (gradient of 0-0.2 degree), which is ideal for storage. Low hydraulic gradient leads to relatively low flow rate. Dillon et al. (2009) suggests having low regional flow rate could ease the recovery phase during MAR.

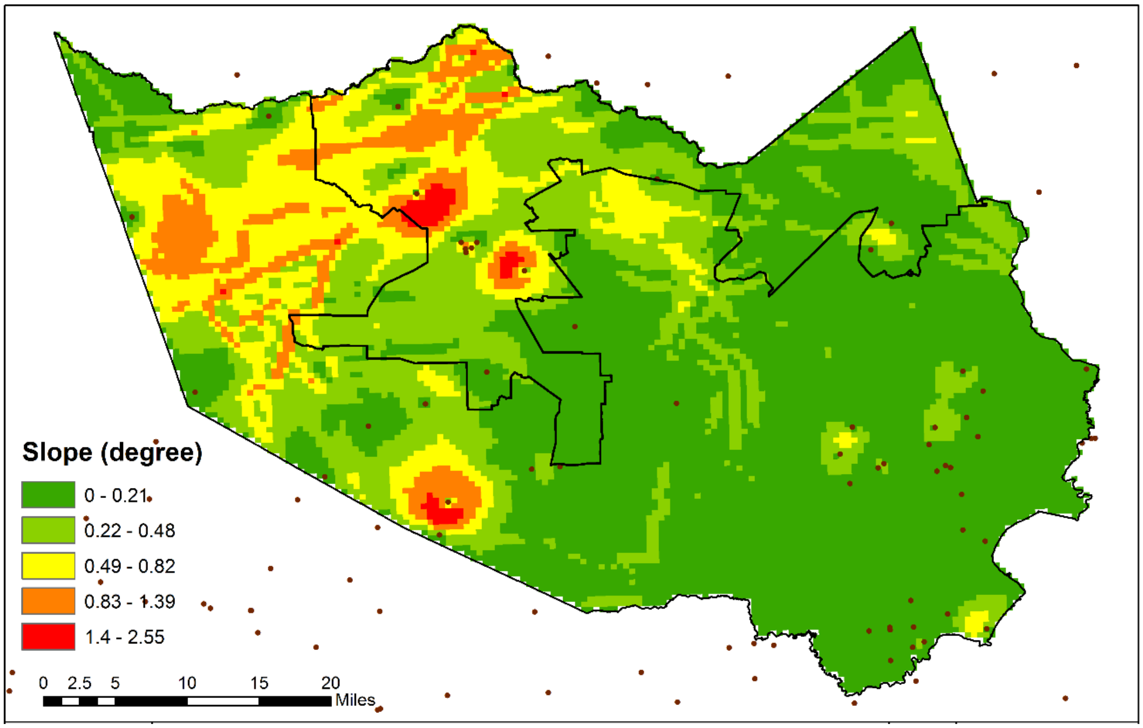

Figure 10. Hydraulic gradient in Chicot Aquifer (Kasmarek et al. 2015)

Aquifer with high transmissivity is favorable to capture lateral flow, otherwise high groundwater mounds would occur and thus lower the infiltration rate (Bouwer 2002). Transmissivity and specific yield are calibrated values extracted from Houston area groundwater availability model (Kasmarek 2012). As shown in the map (Figure 11), central and northern part of the region has moderate to high transmissivity. Southern precinct 4 has medium transmissivity in Chicot Aquifer (ranging from about 6800-1500 ft^2/d), which is ideal since high transmissivity may impede storing water for a significant amount of time, while low transmissivity may cause high mounding during injection. In a study in southern Florida, Brown et al. (2005) ranked transmissivity of 5000-25,000 ft^2/d as the best. Specific yield is a term describing how much water entering the aquifer or being released from aquifer as water level changes. Higher specific yield indicates greater storage capacity. In Harris County (Figure 12), higher specific yield occurs on northwestern edge with value over 0.1. The majority of the region has the value ranging from 0.0005 to 0.006.

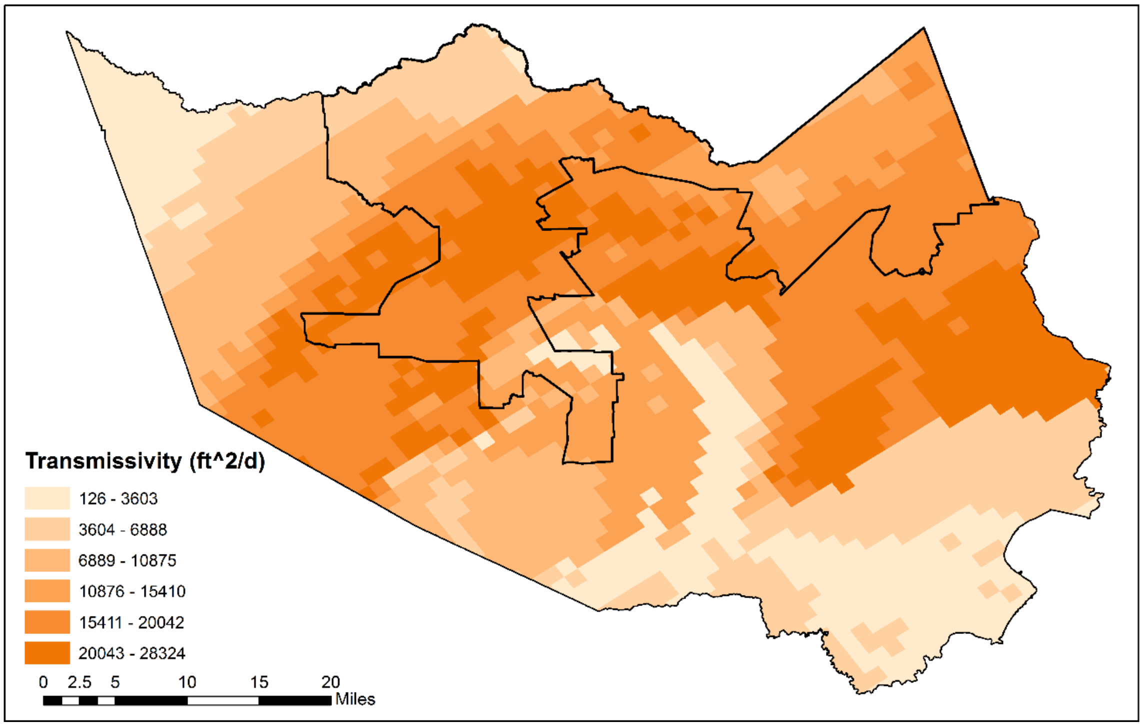

Figure 11. Tramissivity in Chicot Aquifer (Kasmarek 2012)

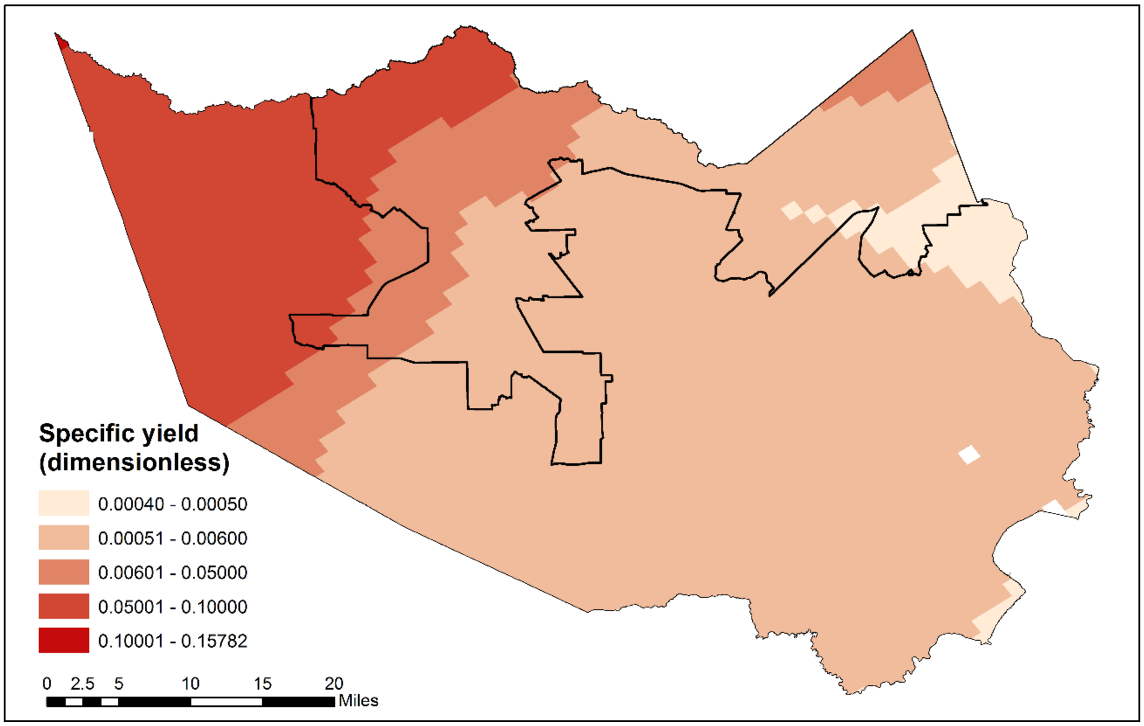

Figure 12. Specific yield in Chicot Aquifer (Kasmarek 2012)

2.3 Land Characteristics

Measured subsidence from 1915 to 2001 is from USGS report (Mace et al. 2006); average annual rate of vertical movement from 2011 to 2016 was extracted from Harris Galveston Subsidence District map (HGSD 2016), data is the estimated annual rate based on active GPS data.

Due to increased water demand came from growing population in the Houston Metropolitan area, water levels in the Chicot declined by up to 200 feet 1943 to 1977 (Mace et al. 2006). Historical overdraft of groundwater has caused significant land subsidence, as high as 9 feet in southeastern Harris County from 1906 to 1978 (Figure 13). As the county shifted to use more surface water in mid-1970s, groundwater withdrawals in Harris County and Galveston

County were reduced from 456 million gallons per day in 1976 to 245 million gallons per day in 2004 (Mace et al. 2006). After the switchover, subsidence has mostly ceased in the southeastern portions of the county, but continues in midwest part, which experienced up to 3.5 ft of change from 1978 to 1995.

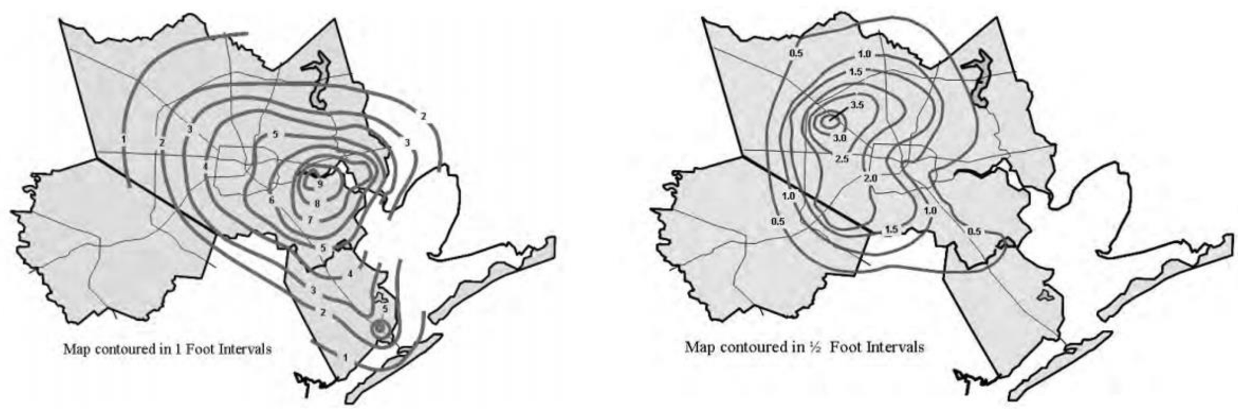

Figure 13. Land subsidence in Harris County. (Left: 1906 to 1978; right: 1978 to 1995) (Reprinted from Mace et al. 2006)

In the map of average annual vertical movement (Figure 14), the red area, located in the western and northern portions of the county, represents higher land subsidence rate. MAR may help relieve the rate of land vertical movement. Land subsidence varies in south precinct 4, while north of the region suffered higher land subsidence.

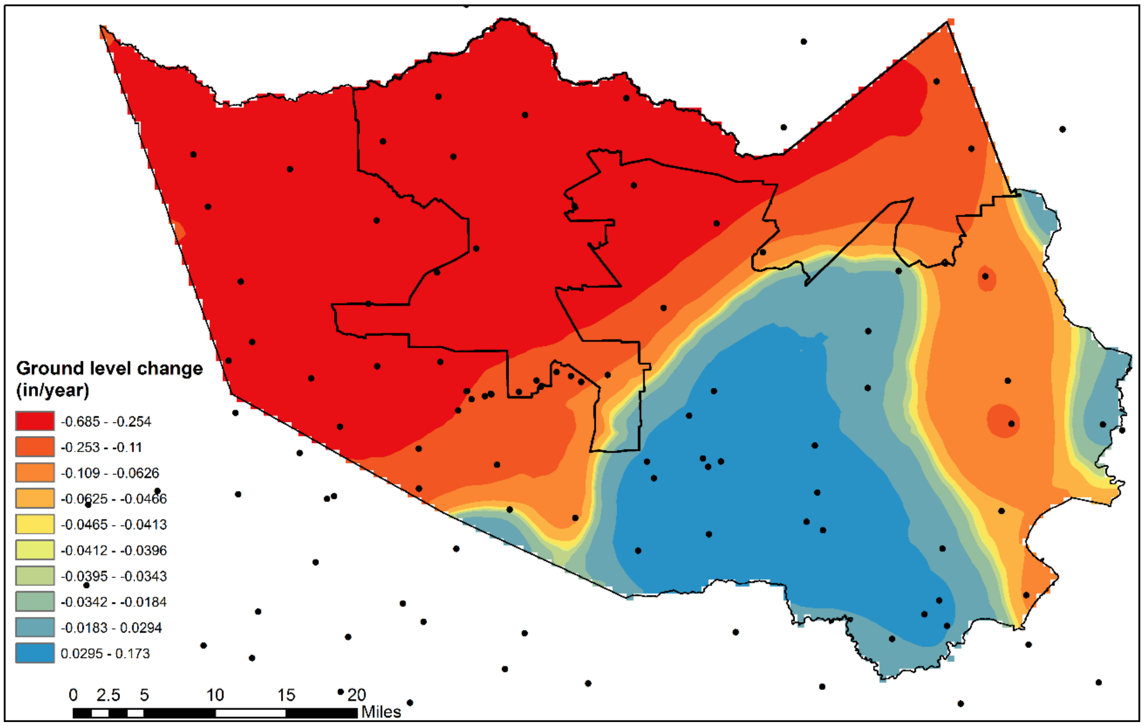

Figure 14. Average annual rate of vertical movement (2011-2016) (HGSD 2016)

Following LID, which aims to reduce runoff near the source, dry wells can be placed near existing detention basins. Detention basins can capture large amount of storm runoff to offer the target recharge volume. Additionally, detention basins could relieve sediment load in storm runoff as well as provide pretreatment of injected water. Detention basins typically have sufficient surrounding areas to allow for dry wells construction. Location of existing detention basins was from HCFCD (Figure 15).

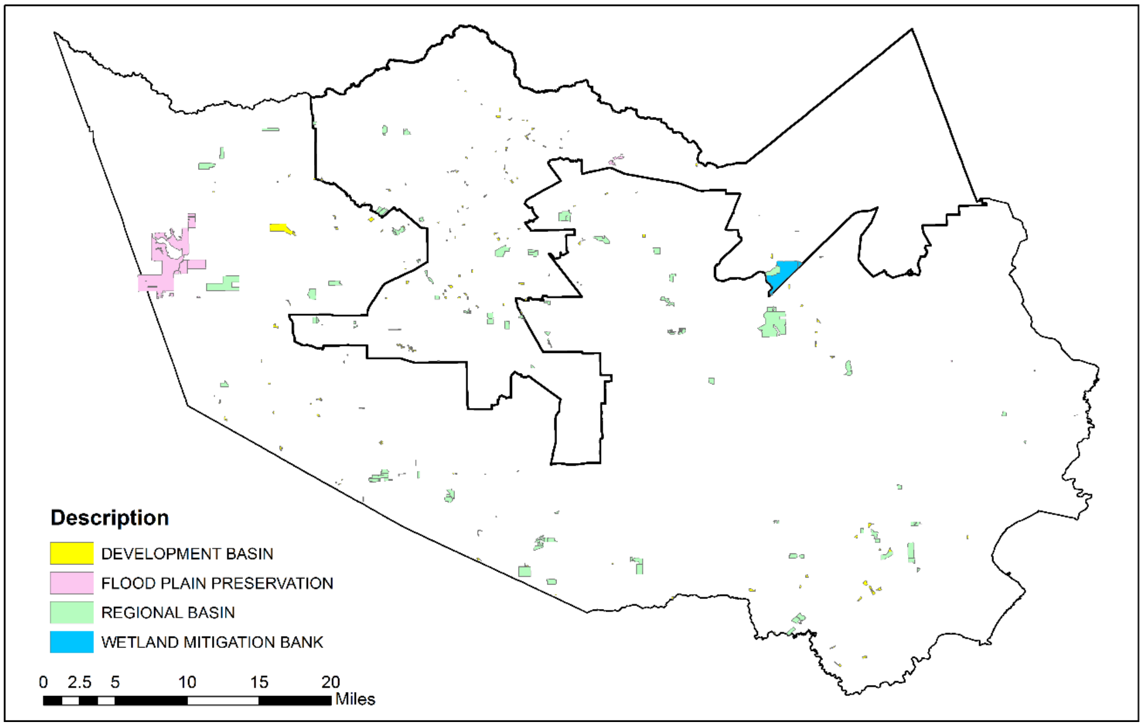

Figure 15. Location of existing detention basins (HCFCD-BMP 2017)

2.4 Water Source and Water Quality

The mapped source water includes watershed boundaries (Figure 16), and 100-year flood plains (Figure 17), all available on HCFCD website (HCFCD-BMP 2017). The floodplain map highlights the areas under high risk of suffering damages in large storm event, while MAR could be applied in those regions to mitigate flood.

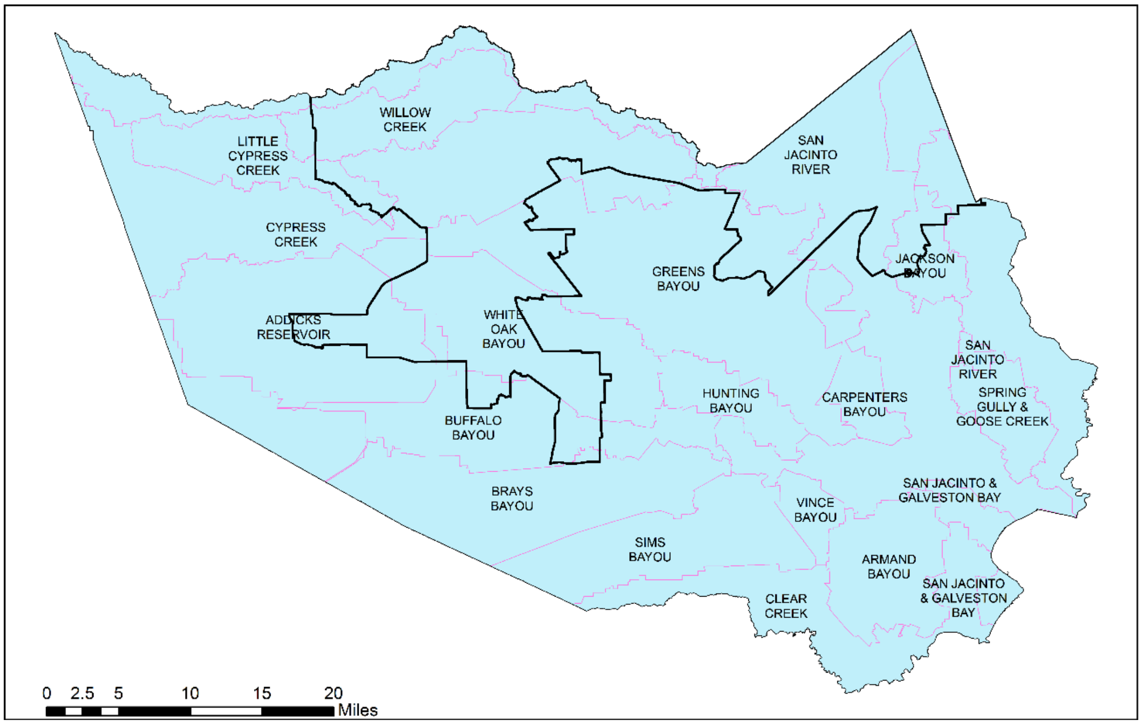

Figure 16. HCFCD watershed boundaries (HCFCD-BMP 2017)

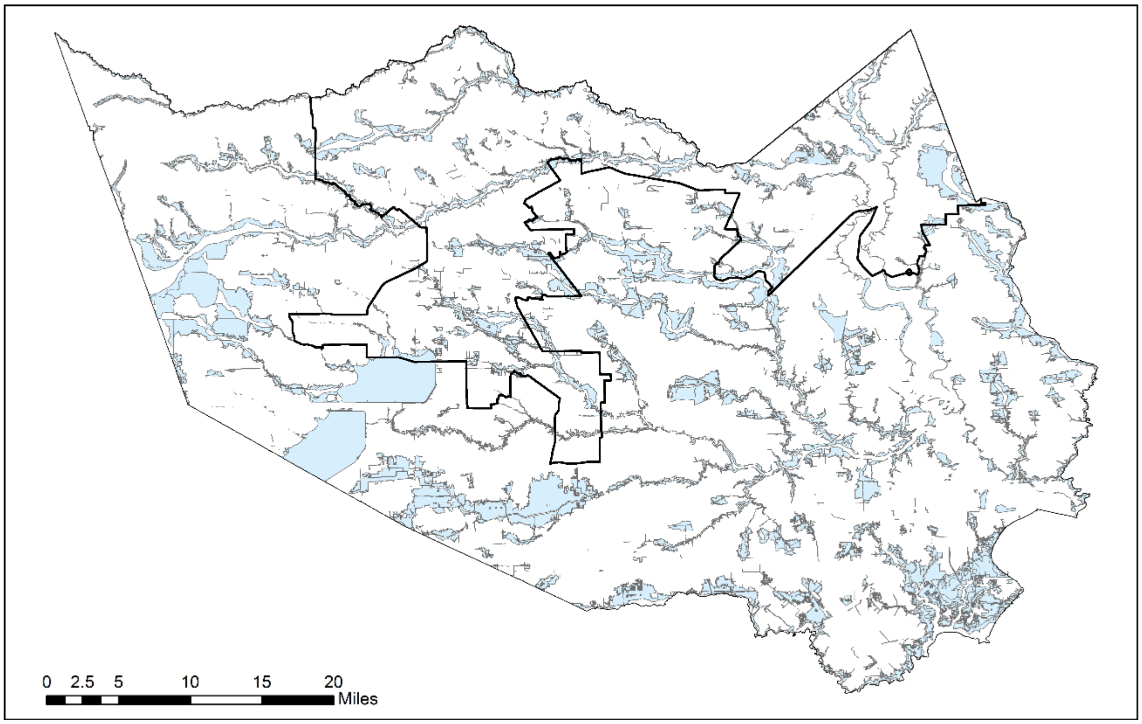

Figure 17. HCFCD 100-year floodplains (HCFCD-BMP 2017)

In terms of water quality, impaired waterways were mapped based on EPA 303d list (Figure 18) (EPA 2017). Blues lines indicate major streams, while impaired water is shown in red. As storm runoff flows through land, the water quality of runoff may be deteriorated by impaired streams. In southern part of precinct 4, the streams are not identified as impaired.

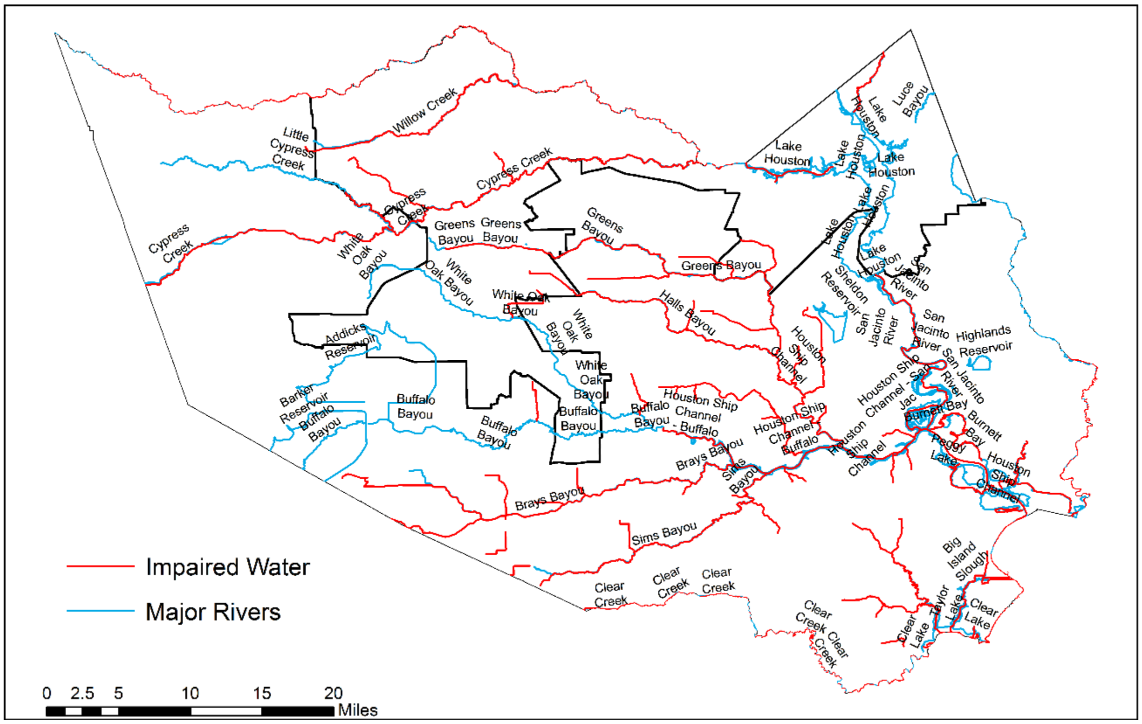

Figure 18. Impaired waterways (EPA 2017)

CHAPTER III

NUMERICAL MODELING OF DRY WELLS FOR STORMWATER MANAGEMENT AND MANAGED AQUIFER RECHARGE

3.1 Synopsis

Dry wells, or stormwater drainage wells, represent a method of managed aquifer recharge (MAR) that is particularly suited to managing stormwater within a low-impact development framework. This strategy has multiple benefits; while minimizing impacts to local ecosystems, it can mitigate flooding from small events, help offset groundwater depletion, prevent land subsidence, and aid in preparing for future droughts. Although contaminants carried by storm runoff may enter dry wells and pose a threat to groundwater quality, prior work suggests that dry wells are safe to use if appropriately designed. This study aims to create a numerical modeling framework for assessing dry well performance and guiding regulations, using Harris County, Texas as an example. To do this, theoretical dry wells were simulated in HYDRUS 2D, with total injection volumes, final water table rise, and extent of tracer migration as key performance indicators. The findings suggest that wells should be designed such that their lengths are maximized but that significant separation distance between the water table and the well remains. As anticipated, lower antecedent soil moisture contents promote storage, as do higher hydraulic conductivity values, which have the added drawback of decreasing residence time. In heterogeneous systems, wells should be screened through clay lenses, rather than ending immediately above. Additional contaminant fate and transport testing is recommended for creating a separation distance standard appropriate for Houston’s highly layered, unconsolidated sediments.

3.2 Introduction

Managed aquifer recharge (MAR) is a family of engineering techniques used to promote the movement of surface water into subsurface, enhancing storage for future use or raising water levels to promote spring flows. Compared to storage via dams, MAR is able to reduce evapotranspiration losses to near zero, halt land subsidence and seawater intrusion, support groundwater-dependent ecosystems, and improve groundwater quality through porous media filtration (Bouwer 2002, Dillion 2009).   MAR may be accomplished via the use of dry wells used to recharge excess stormwater.

Dry wells are large diameter (0.5 - 1 m) wells dug into the vadose zone and screened using perforated concrete or PVC pipe (Figure 19). They extend to a depth approximately 3-5 m above the seasonal high water table, allowing for the "separation distance" necessary to ensure a sufficient residence time of the injected water and to avoid the development of a high groundwater mound (Binkley et al. 2017). They are typically connected to new or existing stormwater collection infrastructure, possibly including retention basins, settling basins, or vegetated buffers, all of which serve as pretreatment mechanisms.

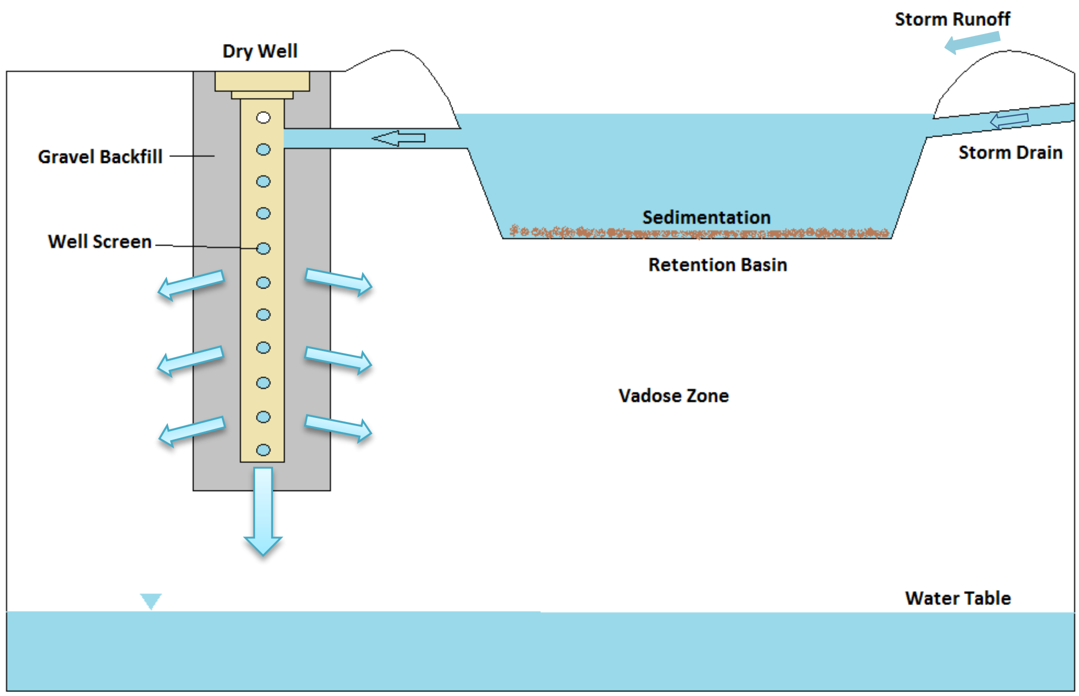

Figure 19. Typical dry well system

Dry wells are considered class V injection wells, which are defined as conduits placing non-hazardous fluid directly to subsurface (Pi et al. 2017). Class V injection wells may only be used to inject high quality water into an aquifer, typically for storage and subsequent recovery. The EPA has no design requirements on dry wells, and dry wells can be operated as long as they are registered with EPA and the operation poses no threat to underground sources of drinking water (Pi et al. 2017). Since urban storm runoff typically carries a suite of potential contaminants, dry wells have the potential to contaminate groundwater. However, additional treatment occurs as water moves through the unsaturated soil media between well and water table, which enhances the contaminant removal via adsorption and biodegradation. In urban areas, the most common pollutants carried by stormwater are oils and grease from vehicles, bacteria such as *E.coli* and fecal coliforms, nutrients (P, NH_4, TKN), and heavy metals (Pb, Hg, Cr, As, Cu) (Pitt et al. 2005). Although high concentrations of contaminants were commonly detected in storm runoff, multiple studies found no evidence indicating that dry wells practices deteriorate groundwater (Wilson et al. 1990, Jurgens et al. 2008, Edwards et al. 2016). Most concluded that with sufficient natural pollutant attenuation and enough separation distance, stormwater injected via dry wells poses no threat to groundwater quality (Edwards et al. 2016).

In the U.S., dry wells are increasingly being adopted as a stormwater management tool, most notably in Oregon, Arizona, and California. Design standards and water quality monitoring requirements vary by state, and typical reflect local conditions such as land use/cover type and underlying hydrogeology. According to New Jersey Stormwater BMP Manual, when planning a dry well, soil characteristics, separation depth between well bottom and groundwater table, sensitivity of the region, and inflow water quality are the key factors to be considered (NJDEP 2016). In California, soil should contain less than 30% clay or less than 40% silt; dry wells should be at least 100 feet away from public supply wells; separation distance should be at least 10 feet; and the site slope should be less than 15% (Pi et al. 2017). Oregon requires a 500 feet setback or 2-year travel-time between dry wells and water supply wells, with a 5-foot separation distance between dry wells and groundwater table (OEHHA 2018). New Jersey specifies a permeability rate between 1 ft/day and 20 ft/day, and a separation distance of 2 feet (NJDEP 2016). In some regions, preexisting dry wells standards have been reformed based on dry well practice (Edwards et al. 2016).

The main goal of this study is to model the performance of dry wells for a range of soil conditions and physical configurations, in order to develop guidelines for its application in Harris County, Texas. Although the simulations presented are specific to its geologic conditions, our overall goal was to develop a modeling framework that can be used to predict recharge volumes and transit times for dry wells in any location.

3.3 Methods

3.3.1 Study Location

Harris County is located in southeastern Texas and contains Houston, the fourth largest metropolitan area in the US. It overlies the northern Texas Gulf Coast Aquifer (GCA), part of the regional Gulf Coast Aquifer System. The GCA is composed of three sub-aquifers (from youngest to oldest): the Chicot, the Evangeline, and the Jasper. The aquifers are composed of laterally discontinuous deposits of gravel, sand, silt, and clay (Kasmarek et al. 2015). Water injected through dry wells will reach the top layers of the Chicot Aquifer, which is made up of a series of surficial alluvium deposits (from youngest to oldest): Beaumont Formation, Montgomery Formation, Bentley Formation, and Willis Formation (Kasmarek 2012). The estimated transmissivity ranges from 3,000 to 50,000 ft^2/day, and storativity ranges from 0.0004 to 0.1 (Carr et al. 1985).

In recent years, Harris County suffered both severe floods and droughts, the most recent of which, associated with Hurricane Harvey. The devastating hurricane started to hit Houston area on August 25th and caused three massive rainfalls in six days. On Sept. 1st when peak rainfall occurred, one-third of Houston was under water (Amadeo 2017). The Cedar Bayou, located in eastern Houston, received a total rainfall of 51.88 inches, setting a record for the largest single storm event in the continental U.S. (Amadeo 2017). For comparison, during the 2011 drought, City of Katy (western Harris County) received a 12-month precipitation of 14.67 inch from October 2010 to September 2011 (Lindner 2012). These events have led the county to explore MAR as a mechanism for buggering these extremes (Binkley et al. 2017).

Within Harris County, depletion of groundwater over 100 years has caused significant land subsidence, an issue MAR could help mitigate. Ground levels have been lowered 9 feet in southeastern Harris County from 1906 to 1978 (Mace et al. 2006). As the county shifted to use more surface water in mid-1970s, subsidence has mostly ceased in the southeastern portions of the county, but continues in mid-western parts, which experienced up to 3.5 ft of change from 1978 to 1995 (Mace et al. 2006). From 2011 to 2016, the average annual rate of subsidence in mid-western parts of the county ranges from 0.2 to 0.6 inch/year (HGSD 2016).

3.3.2 Modeling Methods

HYDRUS 2D was used to simulate water flow in the unsaturated zone and assess the total injection volume, dynamic water table rise, and transit times under a range of dry well scenarios. HYDRUS is a finite element code that uses the Richards Equation to model water movement in vadose zone (Simunek et al. 2012). While it offers several analytical formulations for calculating relative permeability and water retention, we selected the van Genuchten-Mualem equations for use in this study (van Genuchten 1980). HYDRUS can handle prescribed head and flux boundaries, atmospheric condition, constant head and no flux boundaries, which have been useful tools in simulating dry well.

From a previous feasibility analysis, the central portions of Harris County were identified as suitable for the use of dry wells (Binkley et al. 2017). In this application, the County was interested in the installation of dry wells near existing retention basins, to lower construction costs and environmental impact. Three target locations were selected to represent the range of possible subsurface conditions. The saturated hydraulic conductivities (K_{sat}) values tested in the model (0.24 m/d, 0.47 m/d, and 0.78 m/d) were based on soil properties near the selected retention basins. The soil in the unsaturated zone was assumed to be homogeneous and isotropic.

The numerical model domain (Figure 21) was configured with the dry well at the center and extending to 500 m on each side, where constant head boundary conditions are specified. Following common design standards, the modeled dry well was configured to 1 m in diameter, covered by a thin, impermeable layer of concrete on top, and surrounded by 1 m of gravel to facilitate infiltration.

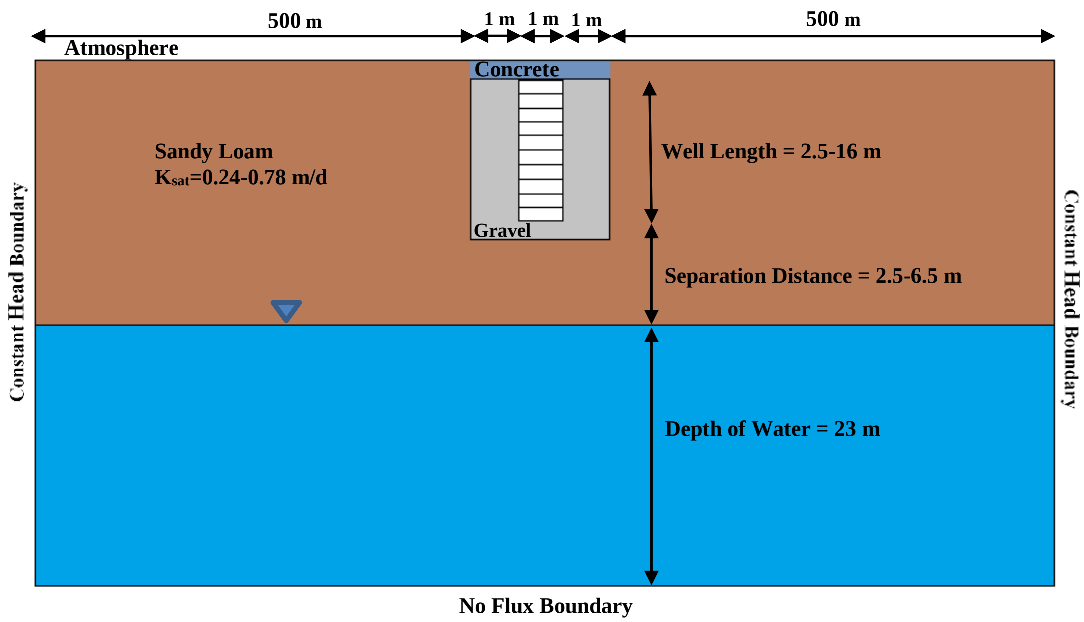

Figure 20. HYDRUS setup

The performance of dry well depends on multiple design methods and local conditions (Section 1.3). Aiming for an extensive investigation on the controlling factors, the well length, duration of injection, separation distance, multiple injections, antecedent soil moisture, K_{sat}, natural recharge, and tracer movement, were addressed in modeling process (Table 1). Well length was determined based on the depth to groundwater table in the target area. The range of other design parameters was based on local standards implemented in other states (Bouwer et al. 2008; City of Portland 2014; Jurgens et al. 2008; OEHHA 2018; Pitt et al. 2012; Wilson et al. 1990). Using these ranges, we created 59 simulations: one baseline scenario and 58 scenarios in which we perturbed the physical factors and design configurations.

Table 1. Modeling Factors

Factors	Description	Range Simulated
Well length	Equals to the static water level maintained in well during injection	2.5 - 16 m
Duration of injection	During which static water level maintained in well	3 hours - 5 days
Separation distance	Distance between well bottom and groundwater table	2.5 m - 6.5 m
Multiple injection	Injection frequency in a 56-day period	Every 7 days, every 14 days
Soil moisture	Antecedent soil moisture ($\theta_r = 0.1,\ \theta_s = 0.39$)	0.11 - 0.35
K_{sat}	K_{sat} of subsurface soil (sandy clay loam)	0.24 - 0.78 m/d
Natural recharge	Rate of natural water infiltration to ground surface	0.05 m/d

In one injection process, the dry well is filled fully with recharge water for a certain time, and recharged water infiltrates into subsurface solely under the influence of gravity. In model simulations, well length was equivalent to the height of water kept in well during injection. Duration of injection is the time when constant water height is maintained in well. The modeled well length ranges from 2.5 m to 16 m; duration of injection ranges from 3 hours to 5 days. When simulating various well lengths and durations, three different soil saturated hydraulic conductivities were also taken into account. In HYDRUS 2D, duration of injection and multiple injection were implemented through time-variable head boundary condition in the well bore.

Separation distance is the distance between well bottom and groundwater level. Local standards usually require a minimum separation distance to ensure enough residence time before the injection joins underground drinking water source. Based on existing standards, values ranging from 2.5 m to 6.5 m were assessed in the model.

In total 7 antecedent soil moisture conditions, model settings of the initial soil water content are presented in Table 2. In unsaturated zone, the soil type is sandy clay loam, which has residual water content Q_r of 0.1, and saturated water content Q_s of 0.39.

Table 2. Antecedent Soil Moisture

Soil Condition	Water Content
Dry	0.11(1.1Q_r) - 0.25 (moderate near water table)
Moderate dry	0.15 - 0.25 (moderate near water table)
Field capacity (FC)	0.22 (FC) - 0.25 (moderate near water table)
Moderate	0.25 (average of Q_r and Q_s)
FC+	0.26
Wet	0.35 (0.9Q_s)
Very wet	0.353

The subsurface soil was assumed to be homogeneous when physical and design factors were assessed, although existence of clay lenses may impede injection movement. To address the subsurface heterogeneity, a thin clay lens was placed at each of four locations: upper well, lower well, well bottom, and above water table. The graph showing locations of clay lens can be found in Appendix. Each scenario contains a 0.5 m thick clay lens, extends 10 m horizontally on both sides of dry well. The clay lens was set to be sandy clay loam with K_{sat} of 0.2 m/d.

Natural recharge is in the form of rainfall infiltrating downwards from top soil surface. According to Harris County Flood Warning System, the average natural recharge rate in Harris County is 0.005 m/d (HCFWS 2018). Since the rate of 0.005 m/d is negligible compared to artificial recharge, to assess effect of natural recharge conservatively, rate of 0.05 m/d was applied in the model. In HYDRUS, the top boundary condition was set to be precipitation of 0-0.05 m/d during injection process.

In baseline scenario, a water column of 2.5 m is maintained in well for 2 days, and then the water is left to drain freely through both side openings and bottom of well. The subsurface consists of sandy clay loam with K_{sat} of 0.78 m/d. Antecedent soil moisture in unsaturated zone is distributed linearly based on the distance to water table. The separation distance between the bottom of a dry well and water table is set to 3.5 m, making a total length of 6 m.

In model outputs, the term, water table rise, represents the change in height of the piezometeric surface, compared to original water table, directly under the well two days after water totally drains from well. Water table rise indicates how well water is distributed horizontally after one injection, and high groundwater mound may impede distribution of incoming water on the next injection. The total injection volume is total amount of water stored in subsurface after one recharge attempt. One major output from HYDRUS 2D is flux (L^2/T) across well openings (well side and bottom), detailed description of how to calculate flow volume from flux can be found in Appendix. When the recharge volume in one dry well is maximized, one well can serve as much area as possible to lower the total construction cost and land requirement. Drain time is when continuous injection ceases, the time it takes to allow the entire injected water to drain from the well. When it requires days to drain water, the water mound and high subsurface soil moisture may limit the system capacity during next injection. In HYDRUS, the water level in the well at a certain time depends on user setting, the drain time was set iteratively until mass balance is achieved. To track tracer movement, one flowing particle was place at edge of well bottom before injection, and the moving distance was measured on certain days (e.g. day 2, day 10, and day 30) after water application.

3.4 Results

The baseline scenario yields 32 m^3 of total injection volume (Figure 22). When recharge application ceases, water totally drains from well in 1.2 days, and water table rise at well bottom two days after injection cessation is 1.4 m.

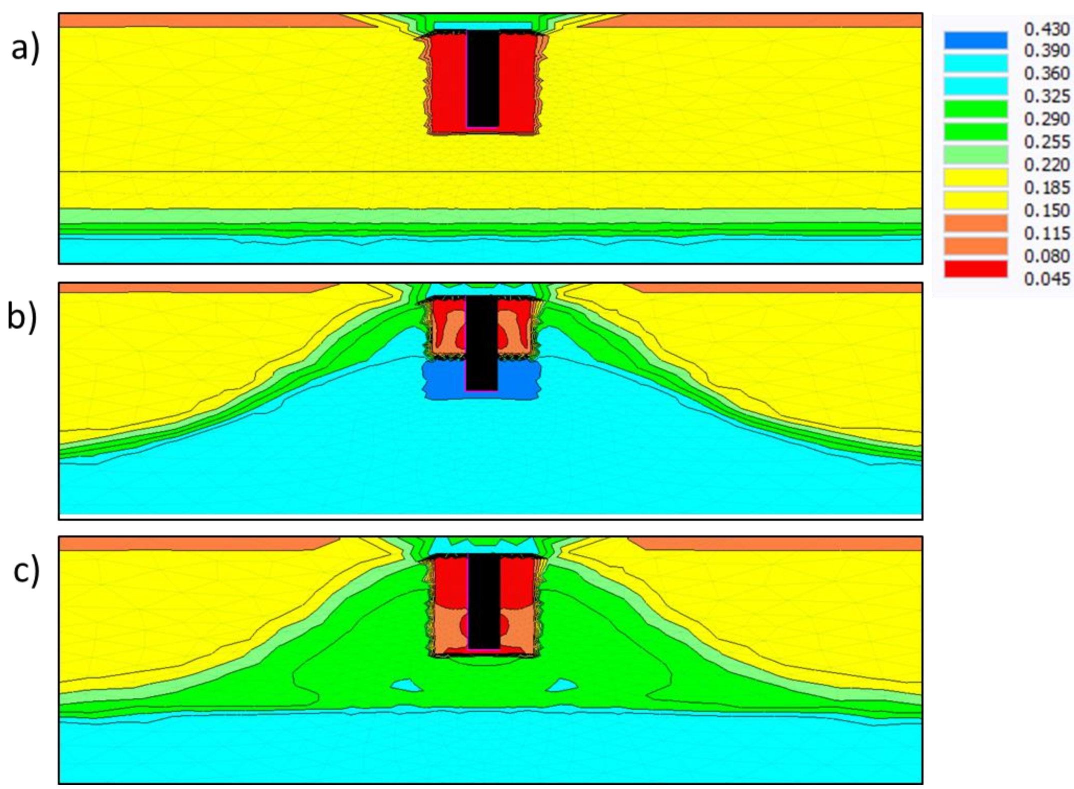

Figure 21. Water content in baseline scenario: (a) initial condition; (b) day 3; (c) day 6

3.4.1 Well Length and Duration of Injection

As shown in Figure 22, total injection volume increases linearly with an extended well length, while the relationship between total volume and duration of injection follows power function. With an extended well length, the water in the well is exposed to a larger unsaturated area, which allows more water transit through well screen entering the unsaturated zone. However, when the same water level is maintained in a well for an extended period (e.g., 3 days), the soil media surrounding the well remains saturated, which lowers the rate of water moving away from well, and thus limits the achievable injection volume. This is reflected by a flatter slope as duration increases from hours to days in Figure 22b. Comparing the two scenarios, extending well length is a more effective way to induce high injection volume. While the long duration of injection should be avoided since the system can be inefficient in distributing water with saturated media surrounding the well.

The recharge differences between three K_{sat} values are consistent in various scenarios. Higher hydraulic conductivity ensures faster movement of water through porous media, and thus allows more water to enter the system.

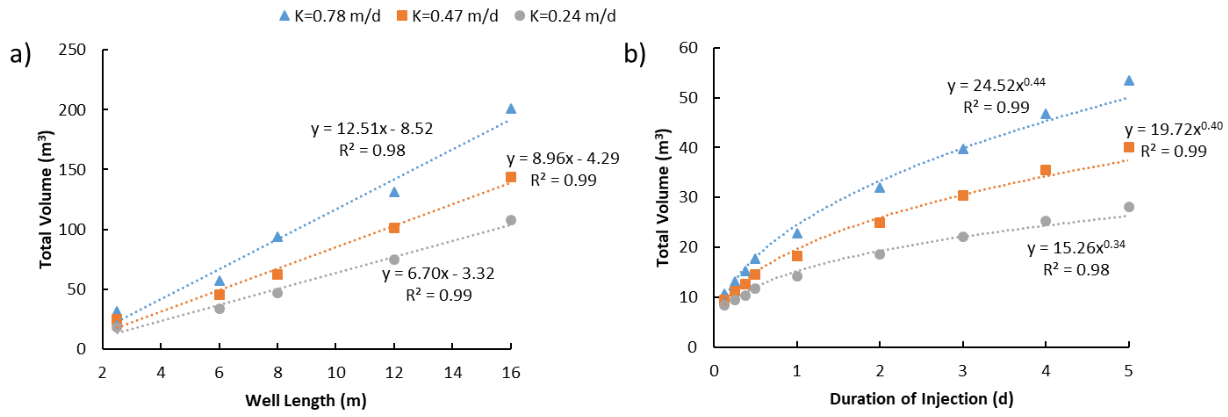

Figure 22. (a) Total injection volume vs. well length, under three K_{sat} soil conditions. The water level maintained in well during injection is equal to the well length; (b) total volume vs. duration of injection (during which the same water level is maintained in well), under three K_{sat} soil conditions

3.4.2 Separation Distance

Higher separation distance (SD) between the groundwater table and the well bottom provides more unsaturated area, allowing the injected water to quickly reach the water table. As suggested by the model (Figure 23), higher separation distance provides a slightly higher injection volume, which in turns creates a higher water table rise. Separation distance of 6.5 m creates 10 m^3 more injection than distance of 2.5 m. On the other hand, water can drain faster with more unsaturated media between well and water table, and the difference in drain time between the SD = 2.5 m and SD = 6.5 m simulations is 19 hours. Although higher separation distance can enhance injection volume, the level of enhancement is less than extended well length. However, higher separation distance ensures better contaminants attenuation. When the total injection volume is of primary importance, a standard separation distance is recommended to save space for placing dry well.

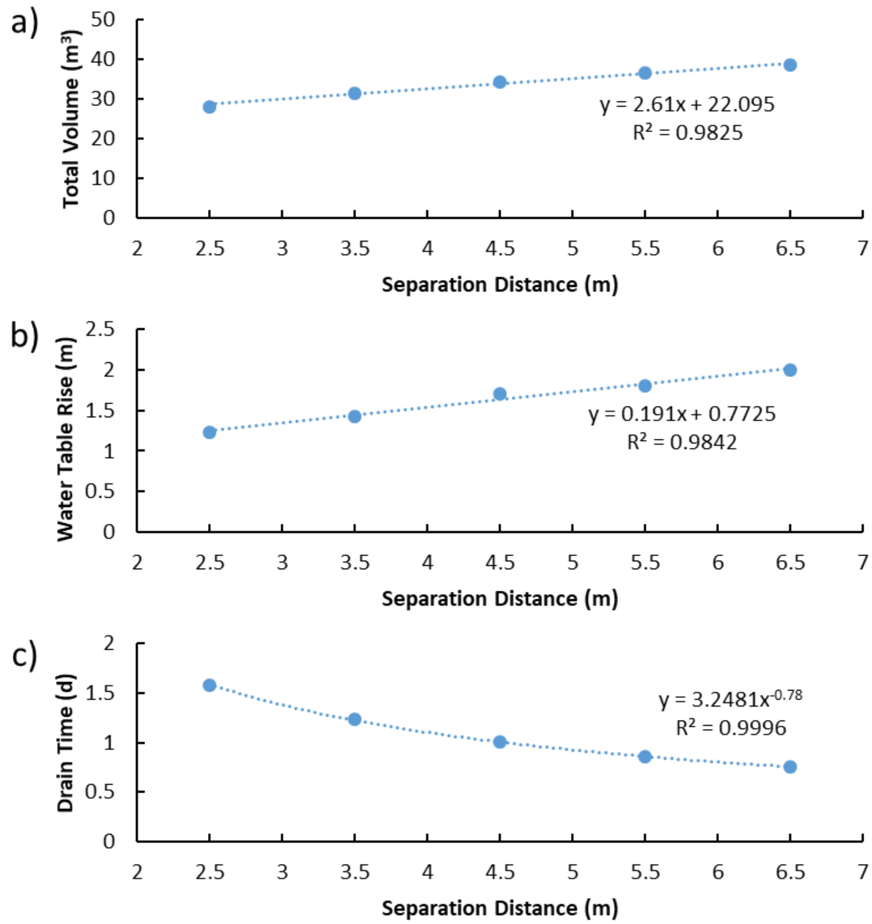

Figure 23. (a) Total volume vs. separation distance; (b) water table rise below the well after two days when injection ceases, the value is the rise with respect to original water table; (c) drain time vs. separation distance. Drain time is when injection ceases, the time it takes to allow water to totally drain from the well

3.4.3 Multiple Injection

In the base scenario, after one injection, the water table mound on day 7 is 1 m, which may interfere with the recharge rate during the subsequent recharge. To assess the impact of the injection frequency, injections occurring at intervals of 7 days and 14 days were simulated for a 56-day injection period. Two K_{sat} values (0.78 m/d and 0.24 m/d) were also included. For the period tested, the relationship between time and cumulative injected volume is linear, and the trend is similar for both higher (Figure 24a) and lower conductivity (Figure 24b) values. The model suggests that during periods of high water availability, such as during summer storms, more frequent recharge can be applied to distribute excessive stormwater into subsurface.

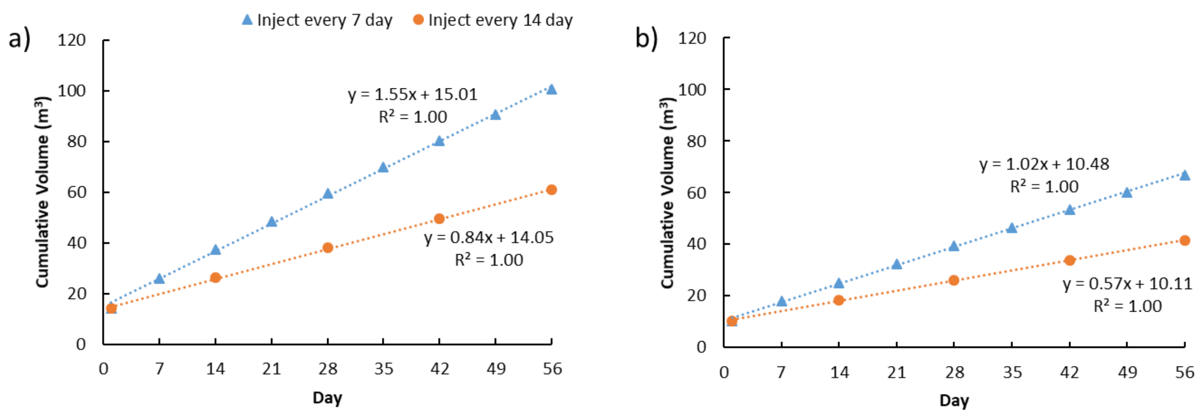

Figure 24. Cumulative recharge volume during a 56-day period for injection frequency of 7 days (blue) and 14 days (orange) for (a) K_{sat} = 0.78 m/d and (b) K_{sat} = 0.24 m/d

3.4.4 Soil Moisture

Antecedent soil moisture is a critical factor affecting the infiltration rate. Seven antecedent soil moisture conditions were specified in model, with the setting described in previous section (3.3.2). When more unsaturated pores present in subsurface, injected water can move through porous media in a faster pace, and thus achieve a higher recharge volume (Figure 25a). The “very wet” soil condition, which has 90% of saturated water content, only allows as half of the injection volume as in moderate dry condition, while requiring a significant amount of drain time (Figure 25a, b). In general, drier soil conditions allow the water table to recover faster (Figure 25c), and thus avoiding a high groundwater mound which could impede water movement. Since dry antecedent soil conditions are favorable to achieving higher injection volumes as well as a lower groundwater mounds, injecting with high subsurface soil moisture (e.g. after a storm event) should be avoided.

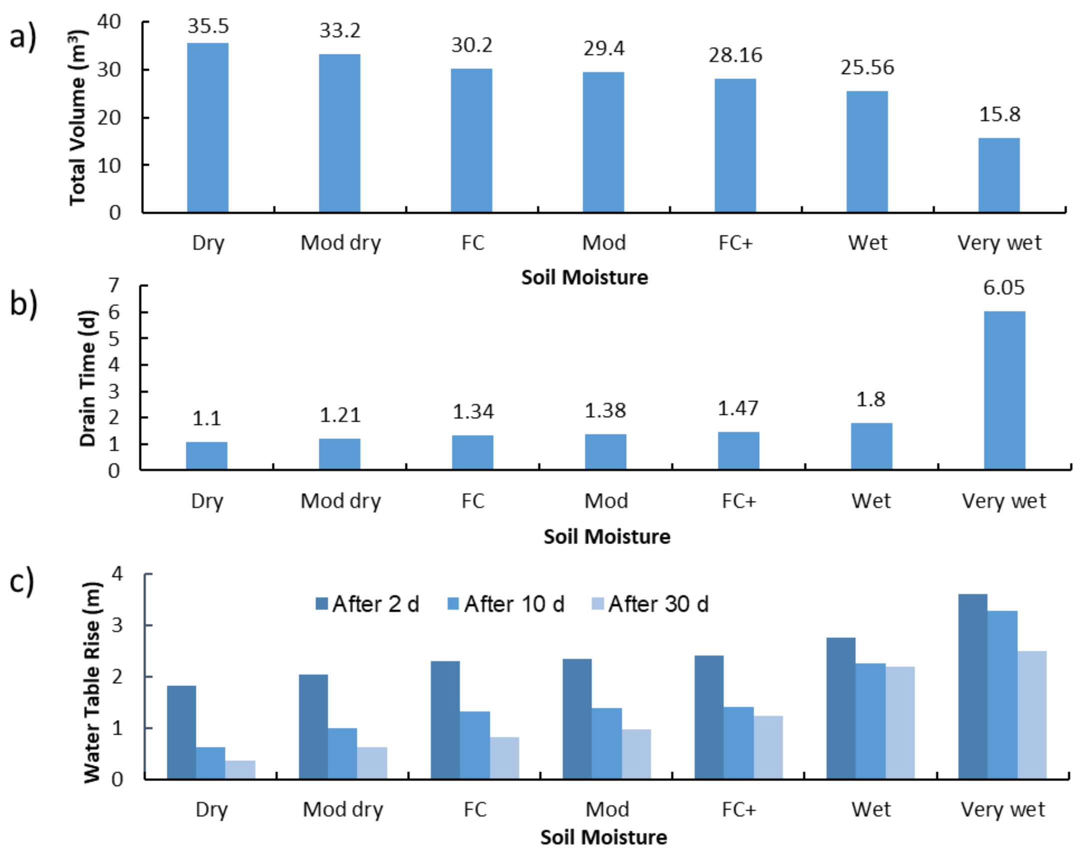

Figure 25. (a) Total volume vs. antecedent soil moisture; (b) drain time vs. soil moisture; (c) water table rise at well bottom, after 2 days, 10 days, and 30 days when injection ceases

3.4.5 Tracer Movement

Although contaminants carried by stormwater can be removed through pretreatment and infiltration process, pollutants can still reach local groundwater and pose threat to potential drinking water source. To model tracer movement, two K_{sat} values and two separation distances were incorporated to compare the impact of the two factors on particle pace. During each injection period, the water level in the well is maintained at 2.5 m for 2 days, and recharge enters subsurface solely under the influence of gravity. The injection process is repeated every 14 days during a 60-day period. Originally the tracer rests at the edge of well bottom.

The results of moving distance suggest that saturated hydraulic conductivity makes a more significant difference in tracer movement than separation distance (Figure 26). Although higher K_{sat} area provides higher recharge rate, the contaminants carried in injection can also be transported fast. In dry well practice, if pretreatment is required, lower K_{sat} areas are more suitable to retard the movement of contaminants and ensure sufficient residence time.

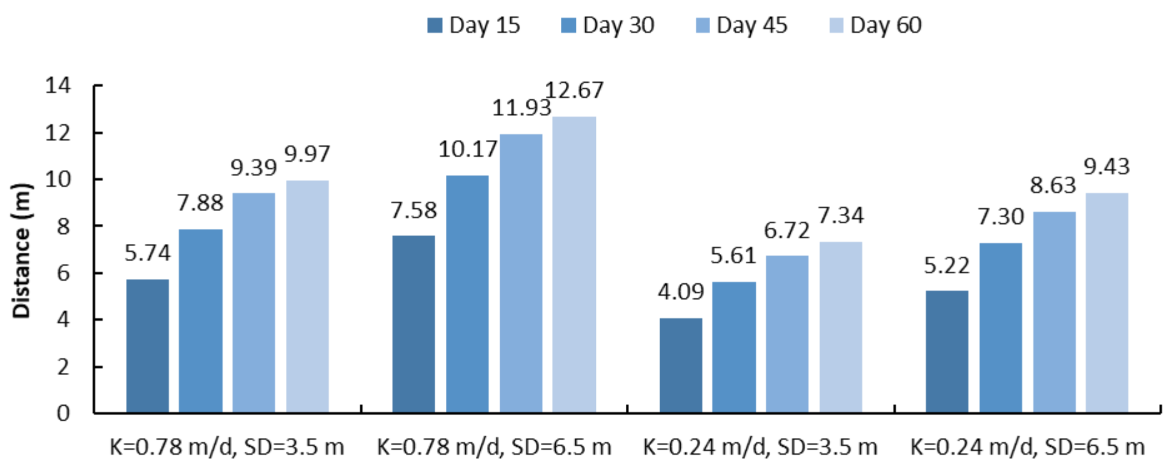

Figure 26. Movements of tracer. The tracer is located at edge of well bottom before injection. Inject every 14 days during a 60-day period

3.4.6 Suburface Heterogenity: Effects of Clay Lenses

When a thin clay layer exists at well bottom, the clay lens successfully blocks around 6 m^3 of water compared to normal condition, and slows the particle pace, while clay lenses at other three locations contribute negligible impact (Figure 27). During dry well construction, when applicable, immediate contact between well bottom and clay layer should be avoided. The model suggests that in order to prevent negative impact from clay layers, the well should be designed so that they are at least 0.1 m below well bottom or are penetrated by the well screen.

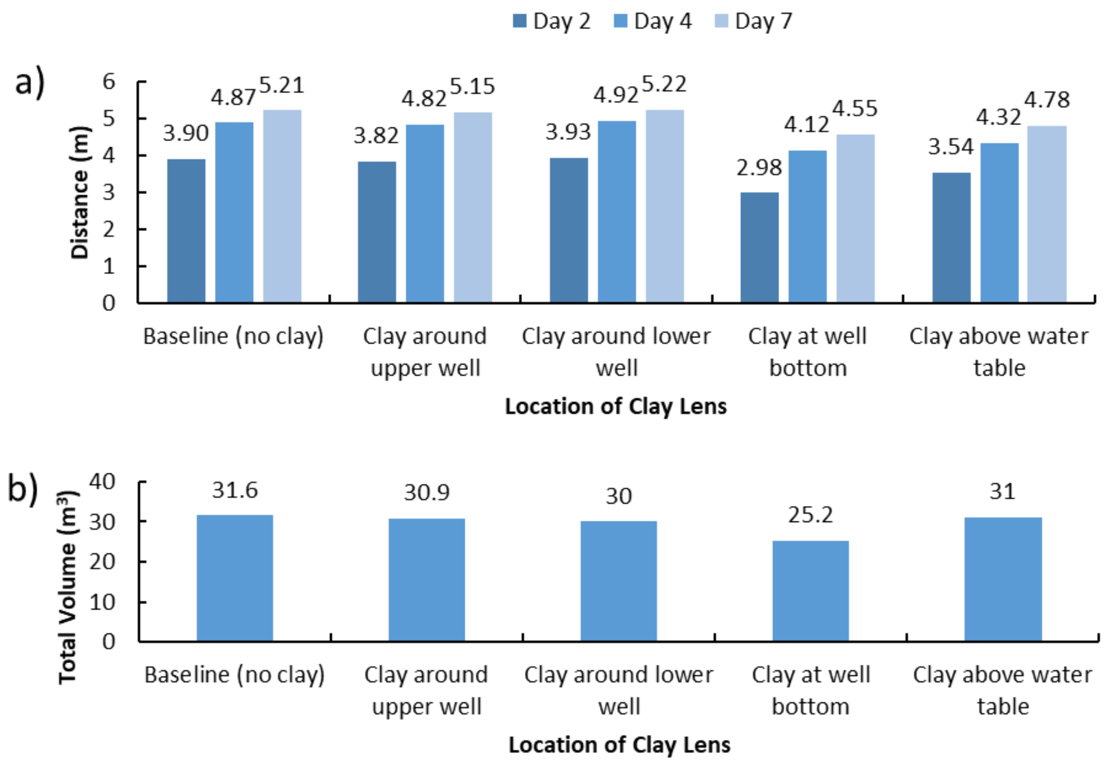

Figure 27. (a) Under various clay locations, particle transport distance on day 2, day 4, and day 7; (b) Total injection volume vs. location of clay lens

3.4.7 Natural Recharge

In model, during a 2-day injection with 0-0.05 m/d of natural recharge on top of domain, natural recharge has no significant impact on neither injection volume nor water table rise (Figure 28). The impact of natural recharge on wider forms of water application can be investigated in future study.

3.5 Discussion

To achieve effective managed recharge, both extended well length and higher separation distance can be utilized. However, based on the model results, extended well length creates a more significant increase in recharge volume if groundwater levels allow such design. Separation distance of 6.5 m creates 10 m^3 more recharge than SD of 2.5 m, while 25 m^3 more water is accepted by subsurface when well length is extended from 2.5 m to 6 m. While determining the design configurations, there is tradeoff between well length and separation distance. Extended well length can be used as a primary method to enhance recharge volume, while separation distance should follow local standard to ensure residence time and efficient purification. This finding confirms that dry wells are most appropriate for regions with deep groundwater tables, if high flow rates are desired.

Another way to achieve recharge volume is through extended duration of injection. When recharge is continued for days, the soil in surrounding area of dry well remains saturated, and hence impede movement of injection. Even though prolonged operation is proved to be an applicable way to enhance injection volume, the recharge efficiency will eventually be limited by saturated soil condition. Long duration of injection, such as over 3 days, should be avoided to ensure the functionality of recharge system.

During storm season, frequent dry well operation may be required to relocate excessive runoff in a timely manner. Based on model results, after one recharge attempt, when the injected water is given days to blend in local water, frequent injection (e.g. in a 7-day interval) would not significantly impede recharge process on the next recharge. Compared to injecting every 14 days, the model suggests injecting every 7 days is achievable to address high runoff volume during surplus period. Further investigation on system sensitivity to injection interval can be performed if more frequent dry well operation is needed in the study area.

The success of dry well application highly depends on local soil conditions. With drier antecedent soil moisture, the soil media provides more unsaturated pores to enhance water movement as well as provide space for storage. When the antecedent soil has moisture of 90% of saturated water content, it takes a significant amount of time for water to drain from well, and the injection volume is half of the volume in moderate dry soil condition. In circumstances where dry wells are located near retention basins and timed operation is possible, we recommend that managed recharge be started several days after the peak storm, to ensure the subsurface soil has enough storage capacity for incoming water.

Although pretreatment facility is commonly incorporated in dry well design, contaminants can still reach subsurface and pose threat to underground drinking water source. Both lower soil K_{sat} and higher separation distance can provide the added strength in impeding particle movement. The model suggests lower K_{sat} makes more difference in slowing contaminants transport. When pretreatment facility is limited, dry well is recommended to be constructed at low K_{sat} (e.g. 0.24 m/d) area to ensure enough residence time. However, subsurface with lower K_{sat} soil would limit the achievable recharge volume, long well length and short separation distance are desired in such area. Different standards on separation distance for various range of K_{sat} is recommended to ensure effective use of subsurface space.

Taking rainfall pattern in Harris County into account, in baseline scenario, which can reduce 30 m^3 runoff in one operation, one dry well can serve 810 m^2 (0.2 acre) of impervious area, for a 2-year rainfall with duration of 30 min. In selected dry well configuration, one dry well takes 7 m^2 surface area. While to reduce 30 m^3 of runoff using bioretention pond, a competitive LID method, 12 m^2 surface area of engineered soil media is required.

3.6 Conclusions

Based on model simulations, following conclusions can be made regarding the performance of a dry well. The extended well length was proved to be the most effective way to enhance recharge volume as aquifer hydrological conditions (depth to groundwater surface and hydraulic conductivity/transmissivity) permits. This finding suggests dry well is more suitable in area with deep groundwater table when recharge volume is of primary concern. High antecedent soil moisture limits infiltration rate and subsurface storage capacity, especially when soil water content reaches 90% of saturated condition. When dry well is located near retention pond and timed operation is possible, injecting several days after the peak storm is preferred. If dry wells have to be operated frequently in surplus periods, injecting in an interval of 7 days is achievable to reduce runoff in a timely manner. In circumstances when injected water is not given adequate time to blend in groundwater, the high groundwater mound may create negative impact on recharge rate during the subsequent operation. Further investigation on system sensitivity to injection interval can be performed if more frequent practice is desired.

To ensure residence time of potential pollutants, a minimum separation distance is commonly specified in existing standards. In dry well design, there is tradeoff between well length and separation distance. Separation distance is recommended to follow local regulation to ensure sufficient space for dry well. Since the model suggests lower K_{sat} is more effective in impeding particle movement than higher separation distance, different separation distance standards can be assigned to various ranges of K_{sat} area to ensure appropriate usage of subsurface space. Contaminants fate and transport tests are recommended to determine the minimum separation distance in study area.

The model suggests that to prevent negative impact from clay layers, the well should be designed so that clay lens is at least 0.1 m below well bottom or are penetrated by the well screen. Before placing dry well and determine well screen, location and size of subsurface clay lens should be identified. In most of the simulations, the subsurface soil was assumed to be homogeneous and isotropic. Multiple clay lenses and lower clay hydraulic conductivity can be included in future study.

Performance of dry wells can be assessed under extreme conditions. For instance, Hurricane Harvey, which brought massive rainfall several days in Harris County in 2017, could be incorporated in simulations. Questions such as when to perform recharge during a massive rainfall, do we need emergency dry wells, can be addressed. Simulating potential extreme events would guide the County enact emergency storm water capture and recharge strategy to address future uncertainty.

CHAPTER IV

CONCLUSION

To successfully apply dry well to reduce runoff, high subsurface infiltration rate and sufficient storage capacity are favorable conditions. During feasibility study in Harris County, central portions of the county (White Oak Bayou Watershed) appeared to be suitable for dry well practice. The targeted area, where 100-year floodplain occurs in a compact fashion, has multiple retention ponds in prepare of storm events. Dry wells are recommended to be placed near existing retention basin to lower both construction cost and environmental impact.

When simulating dry well performance under various soil conditions and design configurations, results from HYDRUS 2D suggest extended well length is the most powerful way to achieve high recharge volume. However, well length design is restricted by depth to groundwater table and enforced minimum separation distance between well bottom and water table. When maximizing recharge volume is the principal objective, dry wells are more appropriate for areas with deep groundwater to secure an adequate space for dry well. Although higher K_{sat} (e.g. 0.78 m/d) ensures high infiltration rate, it also induces fast transport of contaminants, and in such case sufficient separation distance is critical to ensure residence time. Contaminants transport study is suggested to the determine the minimum separation distance for various ranges of soil K_{sat}. High soil moisture (e.g. 90% of saturated water content) in subsurface can significantly limit recharge rate as well as storage capacity. If timed operation is possible, we recommend injection to be started several days after peak rainfall. Further investigations, including sensitivity of system performance to very low recharge interval (e.g. recharge every 3 days), and recharge strategy under extreme conditions, can be performed if the County aims to apply dry wells as the primary stormwater management strategy.

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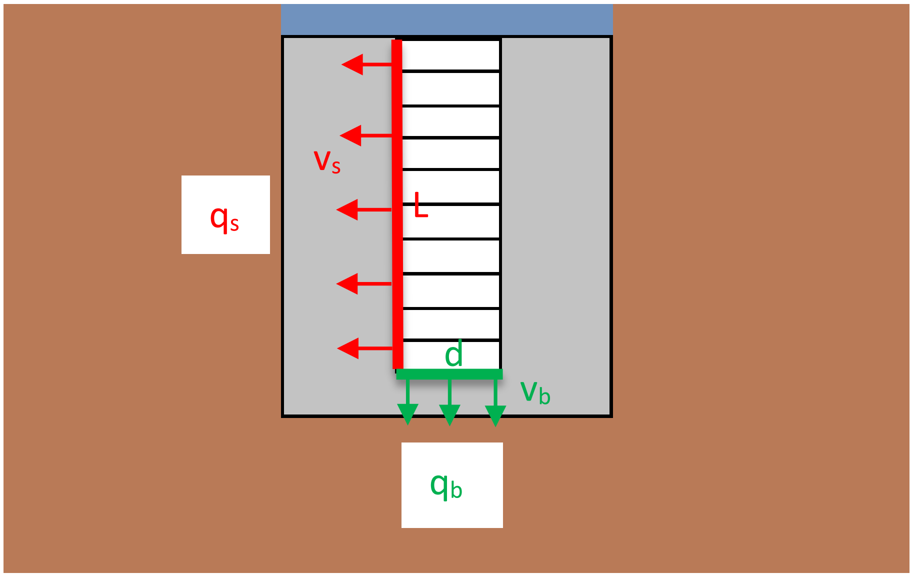
Calculate total volume



Obtain flux from HYDRUS: side of well (q_s), and bottom of well (q_b) [m^2/d]

Bottom flow rate Q_b

$$\mathrm{q_b} = \mathrm{v_b} * d$$

Since well diameter d=1 m,

$$\mathrm{q_b} = v_b$$

$$\mathrm{Q_b} = \mathrm{v_b} A_b = \mathrm{v_b} * \frac{\pi d^2}{4} = v_b \frac{\pi}{4}$$

$$\mathrm{Q_b} = q_b \frac{\pi}{4}$$

Side flow rate Q_s

$$\mathrm{q_s} = \mathrm{v_s} * L$$

$$\mathrm{Q_s} = v_s A_s = v_s * (L * \pi d) = v_s L \pi$$

$$\mathrm{Q_s} = q_s \pi$$

Recharge volume

$$\Delta t = 0.001\ day$$

Calculate Q_s and Q_b at each time step. Then,

$$\Delta V = (Q_s + Q_b) * \Delta t$$

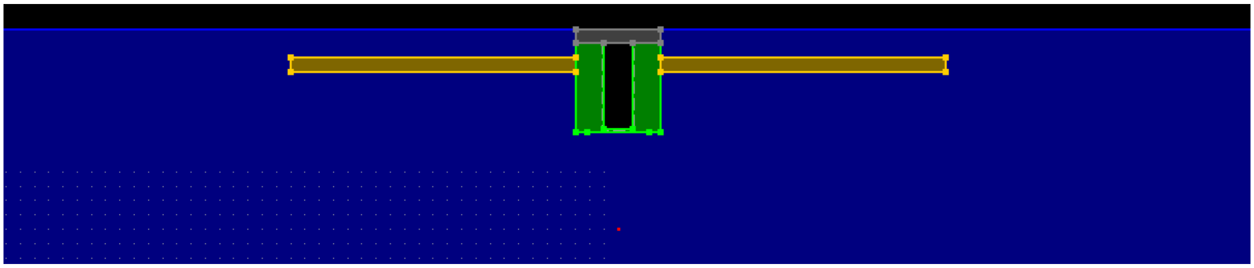
$$V_{total} = \sum \Delta V$$

Clay lens

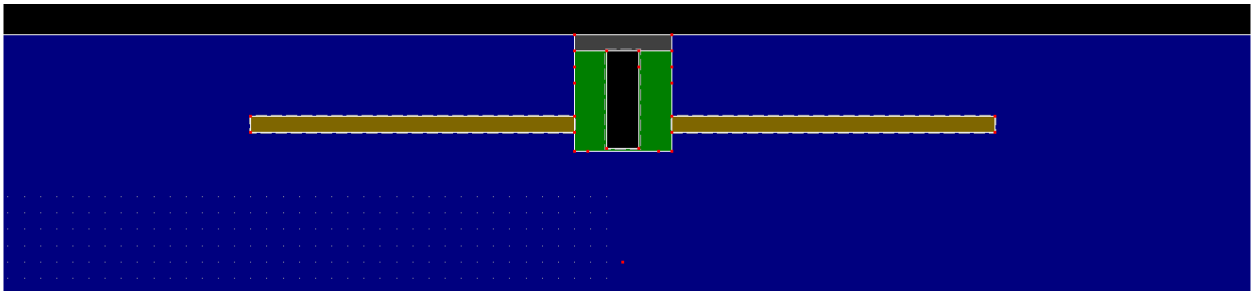
Clay lens locations

Clay lens is 10 m around well, 0.5 m thick. Sandy clay loam, K_{sat}=0.2 m/d.

Upper well:

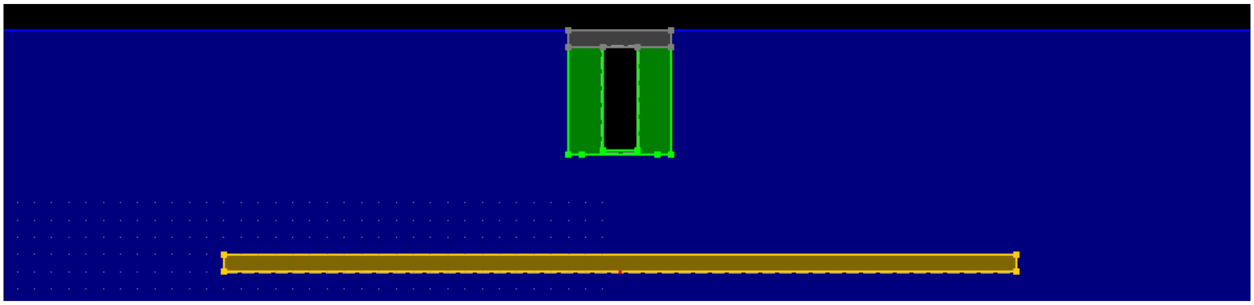


Lower well:



Well bottom:

Above water table:



Seattle dry well discrete sizing standard

Table A1. Discrete sizing for parcel-based projects (Reprinted from http://www.seattle.gov/dpd/cs/groups/pan/@pan/documents/web_informational/p1904279.pdf)

Dry Well Sizing Downstream of Bioretention Sized for Non-SFR GSI to MEF Requirement (91% infiltration) or Permeable Pavement Facility		
Contributing Area (sq ft)	**Dry Well Area (sq ft)**	
	Dry Well Depth = 4 ft	**Dry Well Depth = 6 ft**
500	27	19
1,000	98	67
1,500	164	115
2,000	240	169
2,500	314	222
3,000	390	278
3,500	468	336
4,000	548	396
4,500	630	459
5,000	713	524