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Numerical Evaluation of Community-Scale Aquifer Storage, Transfer and Recovery Technology

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A thesis submitted in partial fulfillment of the requirements for the degree in Master of Engineering Science

NUMERICAL EVALUATION OF COMMUNIITY-SCALE AQUIFER STORAGE, TRANSFER AND RECOVERY TECHNOLOGY

(Thesis format: Integrated Article)

by

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Abstract

Communities in the coastal regions of south-western Bangladesh currently experience severe seasonal water scarcity and groundwater sources of unsuitable salinity. Aquifer storage, transfer and recovery (ASTR), using a seasonal surplus of potable water, is being tested as a potential low-cost (less than $8000 USD) water supply alternative for these communities. A variable-density numerical groundwater model was developed to investigate the engineering technical feasibility of small-scale ASTR systems for the coastal communities in Bangladesh and specifically to support future field site selection and system design. The numerical model was calibrated based on an existing ASTR site and applied to explore the influence of a range of hydrogeological and engineering design parameters. Simulations showed that the water extracted from the ASTR system was able to meet the Bangladesh Drinking Water Standard for Total Dissolved Solids of 1 g/L when injection head or aquifer transmissivity was maximized. A generic ASTR model was developed to examine systems in a non-site-specific context. This analysis showed that four injection wells distributed around a central extraction well with system parameters configured to produce a single injection well plume diameter 1.5 times greater than the level of dispersivity in the system led to high recovery efficiencies regardless of other site characteristics such as injection rate, aquifer depth, and effective porosity.

Keywords

MAR, ASTR, Bangladesh, community-scale, coastal aquifer, variable-density groundwater modelling

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Table of Contents

List of Tables

List of Figures

List of Appendices

Chapter 1

1 INTRODUCTION

1.1 Background

Bangladesh is a low-lying, deltaic country that faces significant drinking water supply issues due to its vulnerability to natural disasters, widespread anthropogenic contamination of water sources and naturally-occurring poor-quality groundwater (e.g. high levels of arsenic, manganese or salt in some areas) (Paudyal, 2002). Year-round access to safe drinking water is a major concern in the south-western coastal area of the country because of frequent storm surge floods and cyclones that contaminate freshwater sources (Karim and Mimura, 2008).

Coastal communities in south-western Bangladesh have traditionally relied on rainwater harvesting during the monsoon season and surface ponds for the remainder of the year as their drinking water source. The aquifers, both shallow and deep, are naturally brackish and therefore are not suitable as a water supply option for the region (Tuinhof, 2011). Surface ponds are repeatedly contaminated by seawater from cyclones or storm surge flooding and cannot be trusted to provide adequate drinking water to coastal communities. Water stored from rainwater harvesting during the monsoon season can only sustain communities for a finite amount of time into the dry season, leaving many communities with critical freshwater shortages during the dry season (Hasan, 2012). There is a need to develop water supply alternatives for these communities which address the seasonal water shortages but which are also cost-effective, resilient to saltwater flooding caused by cyclones and storm surges and are technically feasible to implement (Hasan, 2012).

UNICEF-Bangladesh, the Bangladesh Department of Public Health Engineering (DPHE), Acacia Water (Netherlands) and Dhaka University, Department of Geology are currently evaluating the effectiveness of aquifer storage, transfer and recovery (ASTR) technology in several communities in south-western, coastal Bangladesh. Twenty test sites in the Khulna, Satkhira and Bagerhat districts have been implemented with a proposed scale-up to 100 sites over the next two years. The ASTR schemes being implemented are designed to inject fresh rainwater and/or pond water collected during the monsoon season (May – October) at an approximate total dissolved solids concentration (TDS) of 0.6 g/L into the shallow naturally-brackish (TDS concentration ~5 g/L) sand aquifer via four injection wells, to create a freshwater pocket from which water is extracted throughout the year to meet the needs of the surrounding community (Hasan, 2012; Karim and Mimura, 2008).

The oldest ASTR system installed by UNICEF-Bangladesh and Dhaka University is the Assasuni site in the Satkhira district. In this current study, data from this site was used to develop and test a numerical model of the ASTR system and to conduct sensitivity analyses to evaluate the influence of engineering design parameters and natural geological factors and determine the overall engineering feasibility of the systems. At the Assasuni site, a 13 m thick clay layer overlies a shallow sandy aquifer system which is approximately 11 m thick (Hasan, 2012). This clay layer makes the area ideal for ASTR technology since the clay prevents surface water from infiltrating into the aquifer, potentially causing contamination. The regional groundwater hydraulic gradient, and thus groundwater flows, in the coastal area is minimal (Harvey, 2002; Michael and Voss, 2009). This improves ASTR recovery efficiency since it reduces the potential transport of water downstream from the injection site and the mixing of the injected water with the brackish groundwater (Ward et al., 2009).

1.2 Research Objectives

The first objective of this thesis is to evaluate the technical feasibility of community-based ASTR via rooftop rainwater harvesting and pond water collection as a disaster-resilient water supply technology. Disaster resilient is defined as the ability of the system to recover to expected extraction water TDS concentrations following a flooding event. Systems are considered feasible if water can be made available to the surrounding community for a full 365 days at a suitable standard for TDS (extraction water concentration at 1 g/L or less) and bacteria (aquifer retention time is 2.5 days or greater). The second objective is to advance knowledge of small-scale ASTR systems, their design, operation and effectiveness, since, although this study focuses on ASTR systems in Bangladesh, the results are pertinent to any small-scale ASTR systems installed in similar hydrogeological conditions.

These research objectives were achieved by application of a variable-density numerical groundwater flow model developed in SEAWAT-2005 (Guo and Langevin, 2002) that simulated an ASTR system. The model was used to assess the technical feasibility of using ASTR as a method of storing freshwater for later consumption and use, and to investigate suitable hydrogeological conditions and engineering design options through sensitivity analyses performed on the numerical model. A generic ASTR system was modelled and guidelines have been provided to improve site design from an engineering perspective for future ASTR installations.

1.3 Thesis Outline

This thesis is written in “Integrated Article Format”. A brief description of each chapter is presented below.

Chapter 1 provides background information and states objectives of the study.

Chapter 2 reviews previous research related to managed aquifer recharge (MAR), in particular aquifer storage and recovery (ASR) and aquifer storage transfer and recovery (ASTR), experiments and modelling. A discussion of Bangladesh geology and the need for alternative freshwater supplies for the coastal areas is also provided.

Chapter 3 presents a numerical model used to simulate ASTR systems, sensitivity analyses and generalized guidelines for designing ASTR systems.

Chapter 4 provides conclusions and recommendations for future work and on small-scale ASTR in general.

1.4 References

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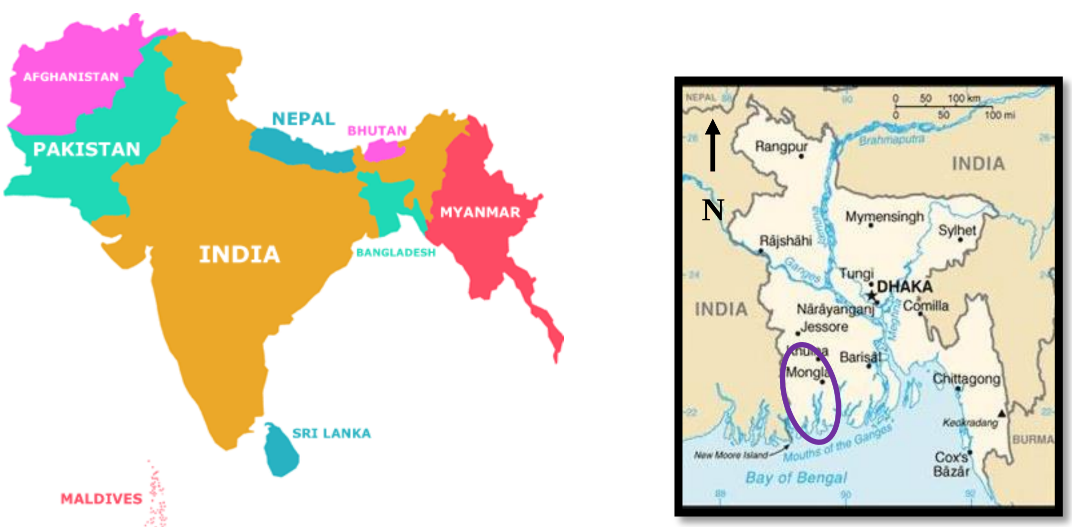
Chapter 2

2 LITERATURE REVIEW

2.1 Introduction

Bangladesh (Figure 1a/b) is often cited as one of the most vulnerable countries to the effects of climate change, particularly sea level rise and the increased frequency of extreme weather events including cyclones (Karim and Mimura, 2008). The south-western coastal region is the most vulnerable area because of its large exposed coast and extensive low-lying land (Karim and Mimura, 2008). Bangladesh is formed on the deltaic plain of three of the world's major rivers: the Ganges, the Brahmaputra and the Meghna and, as such, is low-lying and poorly protected from cyclones and storm surges originating in the Bay of Bengal (Paudyal, 2002). The availability of fresh, potable water in the coastal areas is highly seasonal and dependent on monsoon rains and glacial melt, and is frequently compromised by cyclones and flooding (Karim and Mimura, 2008; Paudyal, 2002; United Nations Development Programme, 2010).



a) b)

Figure 1: a) South Asia map (South Asian Concern, 2013) b) Bangladesh map (study region circled) (U.S. Department of State, 2013).

Traditionally rainwater collected in surface ponds has been the primary source of potable water during the dry season for coastal communities, however, these ponds are frequently contaminated by extreme weather events and therefore are not a reliable source of drinking water for coastal communities (Mallick et al., 2011). For example, following Cyclone Aila in June 2009, the rainwater collected in surface ponds became saline from extensive seawater flooding of the region caused by the failure of coastal embankments (Oxfam, 2013). Other means of storing water, such as dams and reservoirs or cisterns are also not an option because of the frequent seawater flooding that would inundate these supplies, the risk of malaria associated with standing water and the lack of available land space (land is crucial to the survival of many families because it is used for rice fields and shrimp/fish ponds which provide valuable food and income) (Hasan, 2012). Rainwater harvesting carried out during the monsoon season can only sustain communities for a finite period into the dry season, leaving many communities with critical freshwater shortages (Mallick et al., 2011). The groundwater in the south-western coastal area is brackish with some small, isolated, naturally-occurring pockets of fresh groundwater in the deeper aquifer (Hasan, 2012).

Following Cyclone Aila and other cyclones causing similar devastation, non-governmental organizations transported drinking water to coastal communities via road or river (Oxfam, 2013). Four years after Cyclone Aila, surface ponds are again in use but women in some communities still walk up to ten kilometres each day to fetch water from fresh, deep groundwater wells or local residents drink highly brackish groundwater which can have long term health effects (Hasan, 2012; Oxfam, 2013). The World Health Organisation *Guidelines for Drinking-Water Quality* state that water with a total dissolved solids (TDS) concentration less than 1000 mg/L is usually acceptable to consumers (World Health Organisation, 2003). Many residents in the coastal areas consume water with TDS concentrations greater than this standard and the current options to reduce consumption of water with high TDS concentrations are not sustainable and are costly in terms of time or money (Hasan, 2012).

The combination of frequent natural disasters, poor reliability of surface water drinking sources, prevalence of brackish groundwater resources and high population densities creates a significant need in coastal communities for a water supply technology that is resilient to cyclones and storm surges, is cost effective and is available for extraction and human use year-round. Managed aquifer recharge (MAR) is a method of intentionally recharging aquifer(s) to create water stores for future recovery or for improving the local environment (CSIRO, 2010). MAR is frequently used in Australia, the USA, and many European countries. Its use is also increasing in less developed countries including India, South Africa, Mexico and Bangladesh (Page et al., 2010). MAR technology is able to create a viable store of freshwater that is available year-round, making it an ideal option to implement in small communities prone to flooding, land subsidence and cyclones (Dillon et al., 2010; Page et al., 2010). MAR is a suitable option in south-western Bangladesh, in particular, because of the thick geological confining layer overlying much of the region which is able to protect injected freshwater from contamination during saltwater flooding events. Also, the moderate TDS concentrations (1000 – 8000 mg/L) allow small-scale installations to have a large influence on groundwater TDS concentrations (Michael and Voss, 2009).

Aquifer storage, transfer and recovery (ASTR) is a MAR technology where water is injected into the groundwater via a well and extracted some distance from the injection site (Maliva and Missimer, 2010). Few studies have been conducted on the implementation and operation of ASTR systems for small, rural communities (0.3 to 1.5 ML injected per year). Most research has focused on large scale MAR applications (100+ ML injected per year), where injected water has been used to increase groundwater levels, limit saltwater intrusion or the MAR application serves to further treat tertiary-treated wastewater or stormwater for use as a drinking water source. MAR technology has been used in places such as Rajashtan, India, the San Juan River basin in Argentina, and the Tamil Nadu region of India by way of infiltration trenches and basins (Organization of American States, N.D.; Stiefel et al., 2009; United Nations Environment Programme, N.D.) Injection wells have been implemented in towns such as Mehsana, India to raise groundwater levels, but not for pathogen removal accompanied by later extraction and human consumption (Rushton and Phadtare, 1989; Sakthivadivel, N.D.).

This chapter summarizes the literature relevant to the implementation and study of the processes governing MAR technology and examines characteristics of south-western coastal Bangladesh which have contributed to the need for alternative potable water supply options in the area. The MAR literature discussed mainly concerns studies which have been conducted in developed countries and on a large scale. MAR in general, aquifer storage and recovery (ASR) and ASTR technologies, and the factors affecting system efficiency will be discussed in detail. The review focuses on theory, field trials and numerical methods for analysing MAR systems and studies focused on the climatic, geographic and hydrogeologic systems in coastal Bangladesh.

2.2 Water Supply in Coastal Bangladesh

Approximately six million people live in the south-western coastal area of Bangladesh (Bangladesh Ministry of Planning, 2011a; Bangladesh Ministry of Planning, 2011b; Bangladesh Ministry of Planning, 2011c). The entire country lies within the Bengal Basin which is comprised mainly of alluvial and fluvial aquifers formed by the sedimentary deposits of the major rivers of the Ganges, the Brahmaputra and the Meghna, making it one of the largest fluvio-deltaic systems in the world (Michael and Voss, 2009; Paudyal, 2002). Many small rivers also criss-cross over the country, depositing sediments (Paudyal, 2002). The country is low-lying with low regional hydraulic gradients and groundwater flows (Michael and Voss, 2009). The delta is a mixture of fine and coarse-grained layers and in the coastal region in particular, the sandy aquifers are overlain by a thick (1.5m – 232m), clayey-mud geological layer (Hasan, 2012). The stratigraphy of the entire region varies spatially and therefore only local estimates of stratigraphy should be made (Michael and Voss, 2009). Not many hydrogeological studies have been conducted in the region, however, Michael and Voss (2008) suggest that areas can be treated as zonally homogeneous, with their modeling study corresponding well with field data when a homogeneous assumption was made.

Bangladesh is in a major geo-synclinal region and this has caused many of the coastal aquifer systems to become saline, often making them unsuitable for drinking (Karim and Mimura, 2008). Freshwater is plentiful during the monsoon season from May to October, and local residents collect rainwater in ponds and rainwater tanks; however, this supply is finite and many areas are left without sufficient drinking water sources for much of the dry season (Hasan, 2012; Karim and Mimura, 2008).

On average, Bangladesh is struck by one major cyclone each year (Karim and Mimura, 2008). Cyclones cause mass destruction and loss of life and the associated storm surge causing saltwater flooding, which can extend as much as 200 km inland, can have lasting effects on drinking water ponds because of salinization of pond sediments which can contaminate rainwater collected in future years (Hasan, 2012; Karim and Mimura, 2008; Paudyal, 2002). Drinking water sources must be protected from these periodic and catastrophic events. Some low-cost, disaster-resilient solutions that exist to address seasonal water scarcity issues in the region are:

- **Disaster Resilient Ponds -** A modification of the traditional surface pond which incorporates a raised concrete wall around the pond to limit saltwater inundation, however, like a traditional surface pond, the water supply is kept exposed to the air (Figure 2). If flood water levels are high, however, the concrete wall can be breached. Water will also evaporate from these ponds causing significant losses and increasing water salinity over the dry season.

Figure 2: Disaster-resilient pond structure.

- **Tube Wells with Raised Concrete Platforms** – Tube wells are commonly used in Bangladesh (Ravenscliffe, 2013). The raised concrete platform surrounding the opening to the tube well will ensure that saltwater and other contaminants cannot directly enter the underlying aquifer (Figure 3). This option does not, however, address the high TDS concentrations naturally present in the coastal aquifers that make the groundwater water unsuitable for human consumption.

Figure 3: Tube well with raised concrete platform.

- **Pond Water with Slow Sand Filters** – Rainwater is captured in surface ponds during the monsoon season and is passed through a slow-sand filter to remove pathogens and accumulated suspended solids. This technology has been used in several coastal communities, however, lack of public involvement means that the filters are misused and often fall into disrepair and are abandoned (Ravenscliffe, 2013). The pond water can also evaporate by mid to late dry season and the pond and filter water are not protected in the event of saltwater flooding (Ahmed, 2012).
- **Managed Aquifer Recharge (MAR)** – Rainwater is captured during the monsoon season and then artificially recharged into the brackish aquifer (Bangladesh specific scenario) (Dillon et al., 2010). A thick clay layer overlies much of the coastal region making two types of MAR feasible and disaster-resilient (Maliva and Missimer, 2010):
    - **Aquifer Storage and Recovery (ASR)** – Rain water is collected in ponds or on rooftops and is injected into the underlying aquifer during the monsoon season. This creates a pocket of freshwater (impacts on this freshwater pocket, such as mixing caused by dispersion, will be discussed

later) which can later be extracted from the same well as that used for injection (Maliva and Missimer, 2010).

- **Aquifer Storage, Transfer and Recovery (ASTR)** – This method has a similar set-up to ASR, however, injection and extraction take place via different wells with some separation in between (Maliva and Missimer, 2010). This separation allows for subsurface filtration of the injection water which aids in passive pathogen removal, giving this method an advantage over ASR if there is available land space to install separate injection and extraction wells (Pavelic et al., 2004). Passive pathogen removal is valuable in rural communities where people cannot afford other forms of water treatment.

Climate change causing sea level rise and warming waters in the Bay of Bengal will only lead to more frequent and more severe cyclones and storm surge flooding in the future (Ali, 1996). For this reason, suitable technologies must be developed to ensure that there are adequate drinking water sources in the coastal areas. ASTR has the potential to be an appropriate, low-cost and robust option to address this pressing need.

2.3 Managed Aquifer Recharge

Aquifer recharge can be categorized in four ways: natural, which describes how meteoric water reaches the groundwater; unintentional/incidental, such as that caused by leaking pipes, seepage from irrigated areas, or clearing of deep-rooted vegetation thereby reducing transpiration; unmanaged, such as that caused by drainage from stormwater wells or septic tanks; and managed, which is the planned recharge of the aquifer via wells, basins, or trenches (Bouwer, 2002; Dillon et al., 2009). MAR has many applications for both environmental and human benefit. It can be used to store water from various sources including rainwater, stormwater, groundwater from another aquifer, municipal drinking water, or treated wastewater with the intended purpose of securing water supplies, reducing land subsidence, enhancing groundwater quality or preventing saltwater intrusion (Dillon et al., 2009; Dillon et al., 2010).

MAR can be implemented in a variety of ways; hydrogeological characteristics, land availability and intended end-use dictate which type of MAR application is suitable. Both confined aquifers and unconfined aquifers, with or without impaired groundwater quality, can be used for MAR applications depending on local conditions (Dillon et al., 2009). In a confined aquifer, injection takes place directly into the aquifer so as to bypass any overlying impermeable units (Dillon et al., 2009). When MAR is implemented in an unconfined aquifer, water can be placed on the ground's surface or in trenches and allowed to infiltrate through the vadose zone to the water table (Dillon et al., 2009). Figure 4 illustrates the difference between these two systems (Dillon et al., 2009).

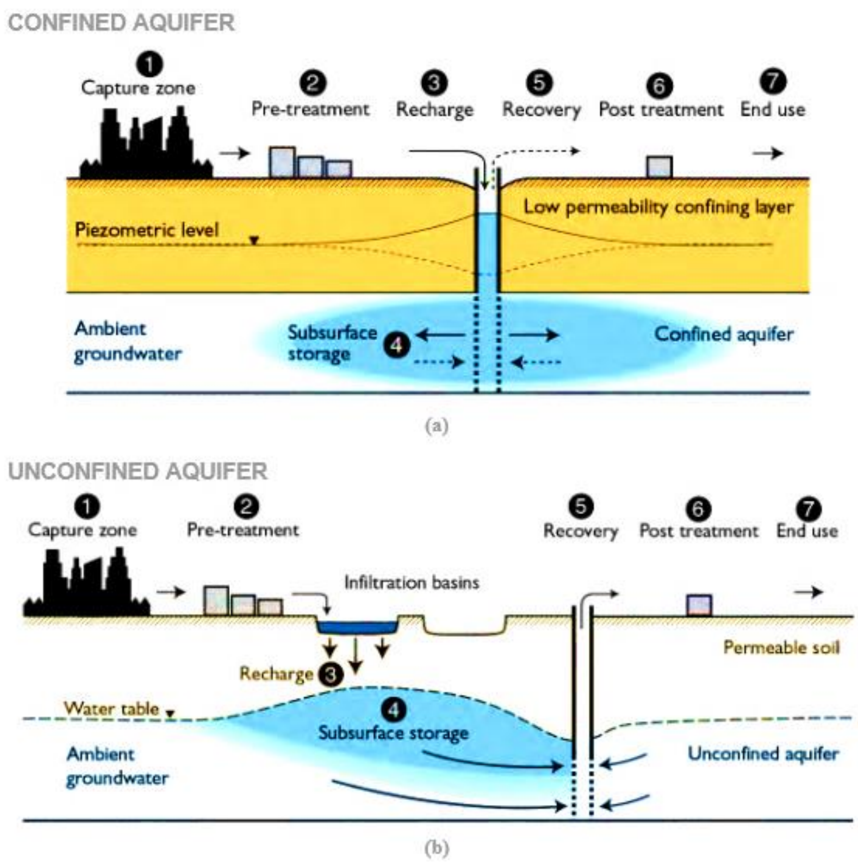


Figure 4: a) MAR in a confined aquifer, b) MAR in an unconfined aquifer (Dillon et al., 2009).

MAR is suitable when water availability is seasonal or irregular – this situation is predicted to become more frequent with climate change (Dillon et al., 2009). Storage of water during times of water abundance means that seasonal fluctuations in drinking water availability can be buffered and water which might otherwise run overland and increase erosion can be stored for later use (Dillon et al., 2009). The storage of water underground also saves on valuable land space and reduces susceptibility to the effects of evaporation or run-off contamination. This makes MAR attractive compared to structures such as dams, as water stresses and land-use requirements increase into the future. Underground storage also has the added advantage of social acceptance, especially when the injected water is tertiary-treated wastewater. Table 1 gives a comparison between aquifer storage and traditional storage of water in dams (Dillon et al., 2009).

Table 1: Storage attributes of MAR versus dams (Dillon et al., 2009).

Attribute	New dams	Aquifer storage
Land area required	large	very small
Proximity to city	far	within
Capital costs	high	low
Investigations costs	high	low
Intake and supply rate	high	low
Evaporation losses	moderate	low
Algal problems	moderate	low
Mosquitoes	moderate	low
Mixing losses	none	none to high
Pathogen removal	some	substantial
Recontamination potential	moderate	none to moderate
Greenhouse gases- embodied energy	high	low
Greenhouse gases- operating energy	low to moderate	moderate
Requirement for viability – presence of:	suitable valley	suitable aquifer

Green colour coding shows the storage type which on average for a given attribute is more favourable or is less constrained.

From an economic perspective, MAR can have many advantages over traditional methods of storing and delivering water, especially in rural areas where water infrastructure is often less developed than in urban centres. When cost and availability of land, cost of required pathogen removal and cost of water transportation are all factored in, MAR is comparable with other engineered systems (Dillon et al., 2009). In disaster-prone areas where water is injected into a confined aquifer, as in the case of the systems implemented in south-western coastal Bangladesh, MAR has added advantages over surface storage combined with slow sand filters because of the resiliency to flooding contamination and the ability to store large quantities of water safely (Hasan, 2012).

MAR does have its disadvantages and cannot be implemented universally. MAR can result in the exposure of high quality water to lower quality groundwater, where the effects of dispersion or hydraulic gradient could render almost all injected water unusable

(Dillon et al., 2009). During times of high need, the aquifer cannot be drawn on at massive rates like surface reservoirs created by dams (Dillon et al., 2009). Higher groundwater flows and changes in water chemistry can also lead to trace element leaching into extraction water (Bouwer, 2002; Vandenbohede et al., 2009).

MAR can include aquifer storage and recovery (ASR), aquifer storage, transfer and recovery (ASTR), infiltration ponds, recharge weirs, infiltration trenches/galleries, dune filtration, bank filtration and several others (Dillon et al., 2009). Figure 5 shows a schematic of various types of MAR technologies.

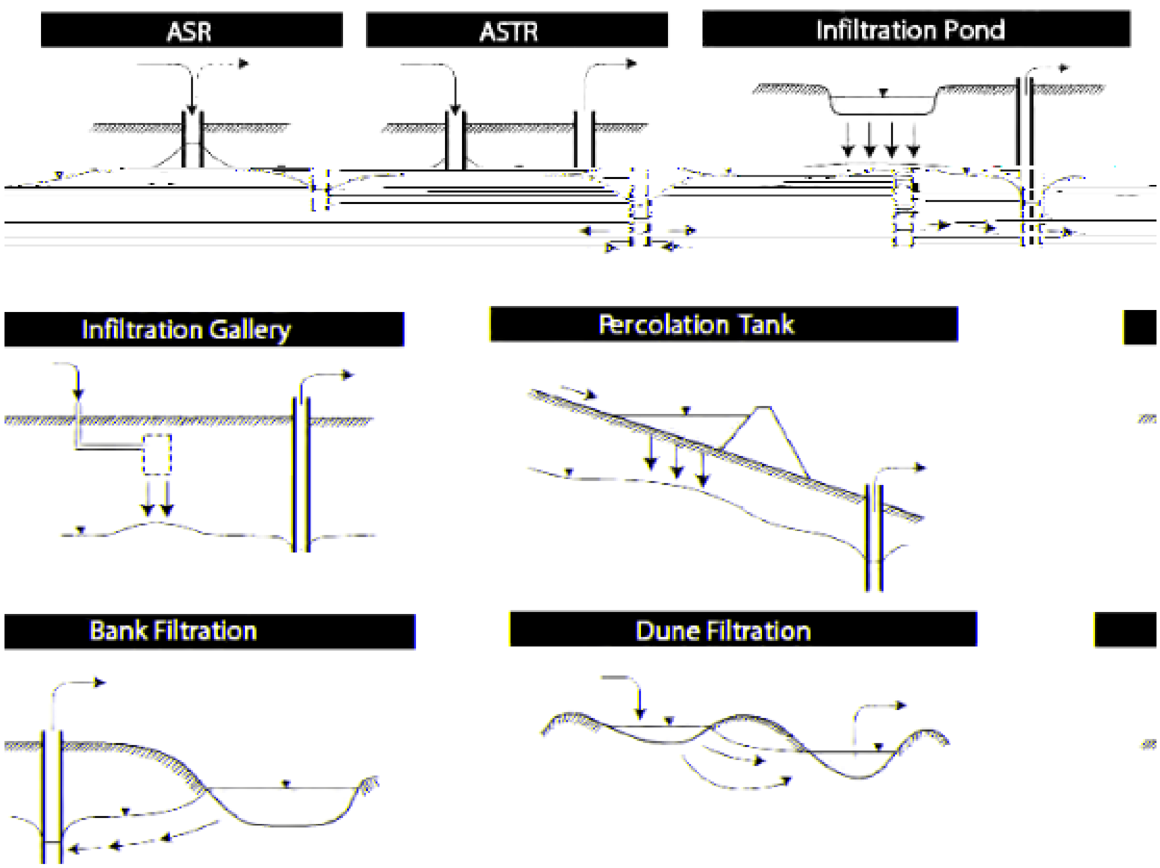


Figure 5: MAR schematics (modified from (Dillon et al., 2009)).

ASR and ASTR are the focus of this literature review. The physics that governs these technologies is similar and therefore conclusions from prior ASR studies are applied to better understand the ASTR system which is the emphasis of this study.

2.3.1 Aquifer Storage and Recovery

Aquifer storage and recovery (ASR) is a MAR technology where surface water is collected and injected directly into an underlying aquifer by way of an engineered system such as an injection well, as a means to raise groundwater levels, improve groundwater quality or create a storage area which can be used for later extraction and consumption (Bouwer, 2002; CSIRO, 2010). ASR has advantages over other MAR technologies because injection occurs directly into a suitable aquifer, allowing for low permeability areas or areas of poorer water quality to be avoided. Land requirements are also smaller than, for instance, a recharge ditch or basin (Maliva and Missimer, 2010). Figure 6 illustrates how an ASR operation works. Specifically for developing countries, the advantages of ASR are that it can fill in gaps in current water supply system availability, it can offset shortages in supply by creating storage capacity, it can ensure that high-quality surface water used for injection is stored safely where if left on the surface it could become impaired from natural disasters such as flooding because of lack of safe storage infrastructure, and the associated cost is often cheaper than alternatives (Almulla et al., 2005).

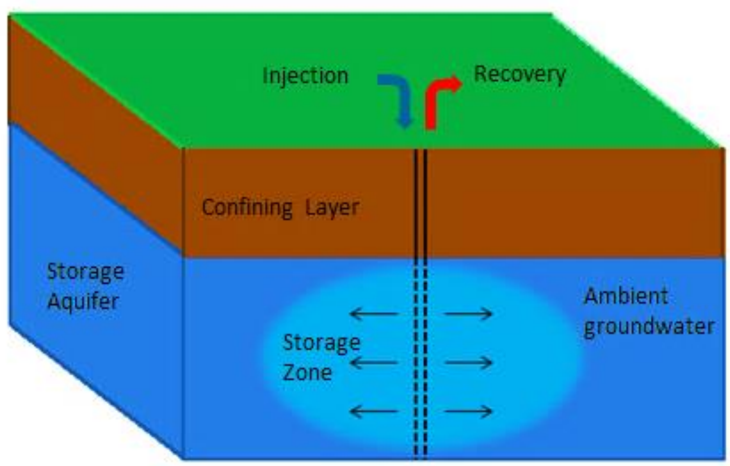


Figure 6: Aquifer storage and recovery schematic.

Introduction of high-quality surface waters into a lower-quality aquifer will result in some losses caused by water quality changes induced by mixing in the injection-water-ambient-groundwater transition zone. Careful evaluation of the benefits and drawbacks of this technology must be made before an ASR system is constructed. For each individual case, the risk of losing all water stores if the water is left on the surface and a disaster event occurs causing contamination of the water must be weighed against the risk of some inherent losses associated with introducing waters of differing qualities to each other.

One of the first instances of freshwater being stored in saline or brackish aquifer was by Cederstrom in 1947 in an aquifer in Virginia (Bakker, 2010). ASR has since been shown to be a promising method to combat saltwater intrusion in coastal aquifers, raise groundwater levels, reduce aquifer salinity and treat tertiary-treated wastewater for use as a drinking water source (Almulla et al., 2005; Vandenbohede et al., 2009). Some examples include the successful application of ASR to reduce saltwater intrusion in the Salalah coastal aquifer in Oman (Shammas (2008) and the Belgian Coastal Plain (Vandenbohede et al. (2009)), and the use of ASR to reduce aquifer salinity in order to restore River Red Gum trees in South Australia (Berens et al. (2009).

The ASR trial conducted by Pavelic et al. (2006a) near Adelaide, Australia involved the injection of freshwater into a brackish aquifer, similar (but a different scale of operation) to the set up applied in south-western Bangladesh. The operation was large with approximately 250 ML of water (TDS $\approx$ 40 mg/L) injected over four years into a brackish, limestone aquifer with a background TDS $\approx$1200 mg/L. Density-induced mixing occurred but this reduced with time as the salinity (density) contrast reduced. The operation successfully created a 25-metre freshwater pocket after four years and was able to achieve an average recovery efficiency of approximately sixty percent (Pavelic et al., 2006a). .

2.3.2 Aquifer Storage, Transfer and Recovery

ASTR, unlike ASR, enables filtration of the injected water through the aquifer sediment, thereby improving pathogen removal (Sidhu et al., 2010). Figure 7 is a schematic of an ASTR system.

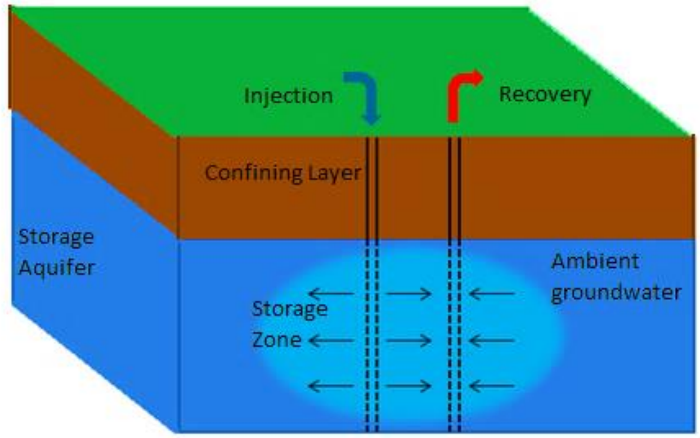


Figure 7: Aquifer storage, transfer and recovery schematic.

While many studies have been undertaken on the feasibility and operation of ASR systems, ASTR studies are limited. Pavelic et al. (2004) numerically investigated the effective well-field design and proper operation procedures for an ASTR system in Salisbury, South Australia. The ambient groundwater TDS concentration at the site was 1900 mg/L while the injectant had an average concentration of 150 mg/L. The maximum allowable extraction TDS concentration was 300 mg/L or a minimum mixing fraction of 0.9 (Pavelic et al., 2004). Mixing fraction (*f*) is defined as the proportion of injection water in the extraction water (Pavelic et al., 2004):

$$f = \frac{C_{amb} - C_{ext}}{C_{amb} - C_{inj}} \qquad \text{Equation 2-1}$$

where C_{amb} is the concentration of the ambient groundwater [M/L^3], C_{ext} is the extracted water concentration [M/L^3], and C_{inj} is the injection water concentration [M/L^3] (Pavelic et al., 2004). The study focused on developing a site-specific optimal well-layout design and therefore cannot be generalized to determine well-layouts at other ASTR (Pavelic et al., 2004). The study established methods to evaluate the system. An effective ASTR system should yield freshwater free of pathogens. The minimum travel time may be calculated to evaluate the effectiveness of the ASTR system in attenuating pathogens (Pavelic et al., 2004). A minimum travel time of at least two to three weeks is acceptable for removal of all pathogens (bacteria and viruses) but an attenuation time of two months is optimal (Pavelic et al., 2004). The equation to calculate the minimum travel time for an ASTR system with one injection and one recovery well is (Pavelic et al., 2004):

$$t_{\min} = \frac{n_e L}{v_{do}} \qquad \text{Equation 2-2}$$

where, t_{min} is the minimum time for injected water to travel from the injection well to the extraction well [T]; n_e is the effective porosity [L^3/L^3]; L is the distance between injection and recovery wells [L]; and v_{do} is the component of flow due to Darcian velocity [L/T] (Pavelic et al., 2004). For study sites where injection water is likely less contaminated, such as where rainwater is used for injection, the minimum travel times is less critical but should still be calculated to ensure that adequate pathogen removal can occur as the presence of pathogens in rainwater can arise from poor collection methods, such as allowing the water to run overland or mixing with other waters.

Brackish aquifers with low dissolved oxygen and an average temperature of 20ºC can be very effective in removing bacteria and other pathogens (Sidhu et al., 2010). In a study by Sidhu et al. (2010), urban stormwater was captured and treated using a combined constructed reedbed-ASTR system, and in situ inactivation of various bacteria and viruses was studied (Sidhu et al., 2010). A test population of bacteria was able to be 90% inactivated by the aquifer in 2.5 days or less leading to the conclusion that ASTR is effective at removing bacteria at levels comparable to engineered treatment. In order to fully remove enteric viruses, however, without requiring long retention times, aquifer treatment should be followed by a treatment such as UV disinfection (Sidhu et al., 2010). In the case of the Bangladesh ASTR systems where bacterial contamination is of greatest concern (enteric viruses have not been detected), a retention time of 2.5 days is recommended to ensure adequate attenuation. If viruses became a concern at a particular site, post-ASTR treatment, such as UV disinfection, is recommended for removal of viruses.

2.4 Factors Affecting ASTR Efficiency

2.4.1 Introduction to Recovery Efficiency

Recovery efficiency (RE) is the percentage of injected water which can be recovered at a suitable standard after one operational cycle (Ward et al., 2009). Suitability is determined based on specific water qualities objectives which are required for the end use. In the case of Bangladesh and many of the other sites examined, suitability depends on an acceptable TDS level (Maliva and Missimer, 2010). RE is the ratio of the volume recovered at a specific quality (V_{rec}) to the volume of water injected (V_{inj}) (Ward et al., 2009):

$$RE = \frac{V_{rec}}{V_{inj}}, (V_{inj} > 0) \qquad \text{Equation 2-3}$$

When freshwater is injected into a saline or brackish aquifer, RE depends on the extent of density-dependent stratification or tilting, the degree of dispersion along the freshwater-saltwater interface, and the ambient groundwater flows (Lowry and Anderson, 2006). A high aquifer hydraulic conductivity may also reduce RE if the injected water is able to migrate beyond the capture zone of the extraction well (Ward et al., 2009). Long storage periods should be avoided as this leads to a reduction in RE as free convection will dominate in the absence of forced convection (caused by pumping), causing potentially large degrees of mixing between the injectant and ambient groundwater depending on the length of the storage period (Lowry and Anderson, 2006). Systems with no storage period will generally have better RE values than those with a storage period (Bakker, 2010).

Numerical modelling and/or field studies should be conducted to determine the suitability of a site for ASTR. Similarly to ASR systems, ASTR systems may not be suitable for all cases and other water storage alternatives might be more appropriate. The cost of some losses associated with the ASTR/ASR storage process must be evaluated against other engineered systems. An ASTR site should not be eliminated simply because 100% RE cannot be achieved. Maliva and Missimer (2010) concluded that RE, especially in the case where freshwater is injected into a saline or brackish water, will almost never be 100% (Maliva and Missimer, 2010).

Bakker (2010) developed an analytical method based on radial Dupuit interface flow to assess the RE of ASR operations in coastal aquifers in the absence of a regional hydraulic gradient. According to Bakker (2010), RE can be determined for ASR systems with or without a storage period, from the dimensionless parameter *D* (Bakker, 2010):

$$D = \frac{Q_{net}}{K_{x,ave}\bar{\alpha}B^2} \qquad \text{Equation 2-4}$$

where Q_{net} is the net flow rate [L^3/T], $K_{x,ave}$ is the average horizontal hydraulic conductivity [L/T], $\bar{\alpha}$ is the density difference ratio defined as $\frac{\rho(C_s)-\rho_o}{\rho_o}$ where ρ_o is the reference density of 1000 kg/m^3 and $\rho(C_s)$ is $\rho_o + 0.7143C_s$ and C_s is the concentration of salt in seawater, 35 g/L, and B is the aquifer thickness [L] (Bakker, 2010; Ward et al., 2009).

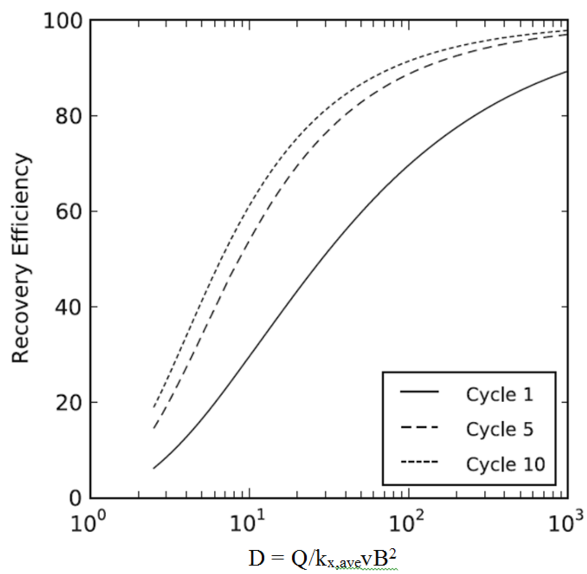


Figure 8: Recovery efficiency as a function of dimensionless *D* (Bakker, 2010).

Figure 8 can be used to estimate RE based on system parameters for various cycles of injection and recovery (Bakker, 2010). These results were compared with the numerical solution developed by Ward et al. (2008) and matched well. Bakker (2010) recommends that when a site is predicted to have an acceptable RE as per Figure 8, the effects of mixing and chemistry on the system should then be considered (Bakker, 2010).

2.4.2 Regional Hydraulic Gradient Effects

Regional hydraulic gradients cause lateral groundwater flow. In the case where freshwater is injected into a brackish or otherwise poor water quality aquifer and a regional hydraulic gradient is present, the plume will migrate downstream under the influence of a lateral flow field, and eventually a point will be reached where the plume can no longer be recovered effectively and only ambient groundwater will be extracted. Where the injection plume and the extraction well capture zone coincide, water of an acceptable water quality can be extracted. The distance from the recovery well to the upstream point of this interaction zone is called $x_{i,upstream}$ (Figure 9) (Ward et al., 2009).

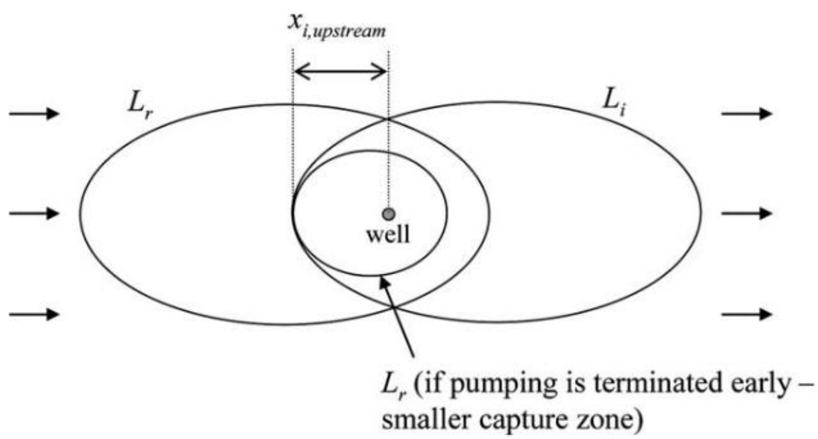


Figure 9: $x_{i,upstream}$ as determined by a lateral flow field (L_i is the locus of the injected plume front and L_r is the locus of the extraction plume front) (Ward et al., 2009).

Ward et al. (2009) developed a dimensionless number to describe the technical viability of an ASR operation under the influence of a regional hydraulic gradient. The technical viability ratio can be described as the ratio of lateral hydraulic conductivity, regional hydraulic gradient and storage time to the porosity and distance to the fresh-salt interface (Ward et al., 2009):

$$R_{TV} = \left| \frac{K_{x,ave} I t_{storage}}{n_e x_{i,upstream}} \right| \quad \text{Equation 2-5}$$

where $K_{x,ave}$ is the average horizontal hydraulic conductivity [L/T], I is the regional hydraulic gradient [-], $t_{storage}$ is the storage time [T], n_e is the effective porosity and $x_{i,upstream}$ is the distance to the fresh-salt interface (Ward et al., 2009). It is recommended that R_{TV} be kept below 1 if a system is to be successful. R_{TV} values > 1 mean that the lateral shift in the freshwater plume is too great and recovered water will mostly be comprised of ambient groundwater (Ward et al., 2009).

2.4.3 Variable Density Groundwater Effects

In an ideal homogeneous aquifer where density differences and mixing are neglected, the interface between a freshwater injected plume and ambient, brackish groundwater is vertical. However, when freshwater is injected into a saline or brackish aquifer, density differences can create an unstable freshwater-saltwater interface and cause macrotilting (Figure 10) (Bakker, 2010). This has significant implications on the RE since pumping will cease when water which does not meet extraction water standards begins to enter the well. This will happen more quickly when there is tilting because ambient groundwater near the bottom of the well, which has been less displaced, will enter the well before ambient ground water at shallower depths. This will mean that a large proportion of fresh water present in the top of the aquifer will not be extracted and there will be reduction in RE (Ward et al., 2009).

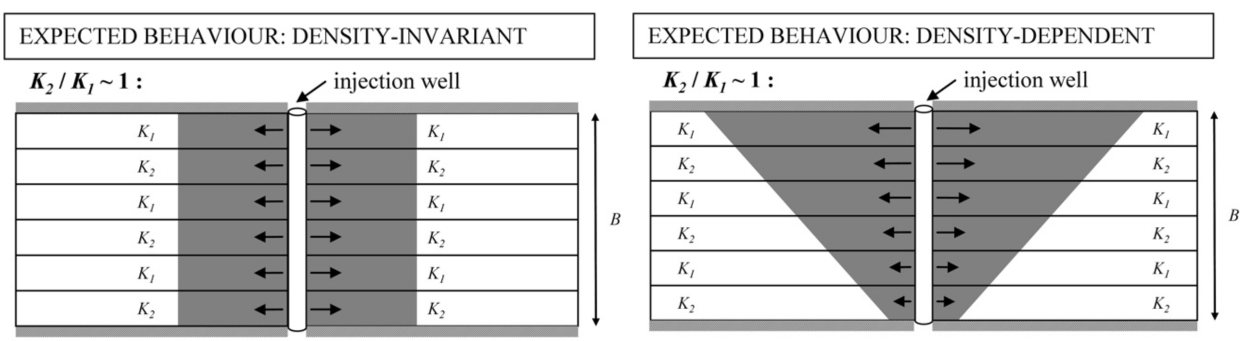


Figure 10: Density-invariant versus density-dependent ASR plumes (Ward et al., 2009).

Ward et al. (2009) numerically studied the effects of the density difference between fresh injection water and saline/brackish ambient groundwater on RE. It was concluded that greater permeability ratios between stratified aquifer layers lead to higher RE values for the system, since areas of lower permeability retard the movement of freshwater upwards

(Ward et al., 2009). The higher the anisotropy ratio ($K_{x,average}/K_{z,average}$) of the aquifer, the higher the RE will be when density differences are present as lower vertical hydraulic conductivity will impede the movement of the freshwater plume due to buoyancy effects (Ward et al., 2008; Ward et al., 2009).

Density effects are most prominent in the storage phase when there is no pumping and thus no forced convection. Free convection during the storage phase occurs solely because of density differences and is exacerbated by slow pumping rates, long storage durations, large hydraulic conductivities, and thick aquifers, even with small density differences (Ward et al., 2007; Ward et al., 2009). Groundwater velocity caused by free convection can be defined as v_{free}:

$$v_{free} = \frac{K_{z,ave}\bar{\alpha}}{n_e} \qquad \text{Equation 2-6}$$

where $K_{z,ave}$ is the vertical hydraulic conductivity [L/T] (Ward et al., 2009).

When freshwater is present in a brackish aquifer during the storage period, tilting will occur because of density differences (Ward et al., 2009). The storage tilt ratio, or R_{ST}, can be described as (Ward et al., 2009):

$$R_{ST} = \frac{K_{z,ave}\bar{\alpha}Bt_{storage}}{n_e\left(x_{i,upstream}\right)^2} \qquad \text{Equation 2-7}$$

where $K_{z,ave}$ is the average vertical hydraulic conductivity [L/T] (Ward et al., 2009). The value of R_{ST} should be < 1 to reduce the amount of density-induced tilting that occurs during storage (Ward et al., 2009).

2.4.4 Dispersion Effects

Dispersion has both positive and negative impacts on ASR/ASTR system efficiency. Dispersion can attenuate some of the density-driven effects by creating a larger salt-freshwater transition (mixing) zone which reduces the density gradient and therefore density-induced flows (Ward et al., 2007). Aquifers with high dispersivity, however, may not be suitable for implementing ASTR systems if it causes injected fresh water to mix too greatly with low-quality ambient water found at the transition zone or in lower permeability zones (Figure 11) (Ward et al., 2009). In Figure 11, higher levels of dispersion in the system will result in a larger transition zone width between the fresh and saltwater, potentially to the point where the extraction well is encompassed by the transition zone area and poor quality water is extracted.

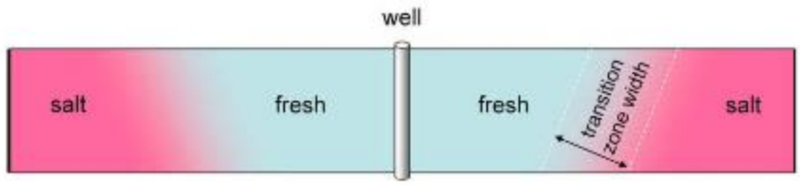


Figure 11: Dispersive mixing causing a freshwater-saltwater transition zone (Ward et al., 2009).

A dimensionless parameter referred to as the relative dispersivity was developed by Ward et al. (2009) to describe the effect of dispersion on the system. This parameter is defined as (Ward et al., 2009):

$$R_{disp} = \frac{\alpha_L}{x_{i,upstream}} \qquad \text{Equation 2-8}$$

where α_L is the longitudinal dispersivity [L]. It was determined by Ward et al. (2009) that when R_{disp} is much less than 1, the effects of dispersion on the system efficiency will be minimal, whereas if R_{disp} is equal to approximately 1 or greater, achieving an acceptable RE could prove difficult.

Dispersivity is difficult to estimate at field sites without conducting tracer tests. In the absence of field data, review studies such as Gelhar et al. (1992) provide the best estimates of dispersivity (Figure 12).

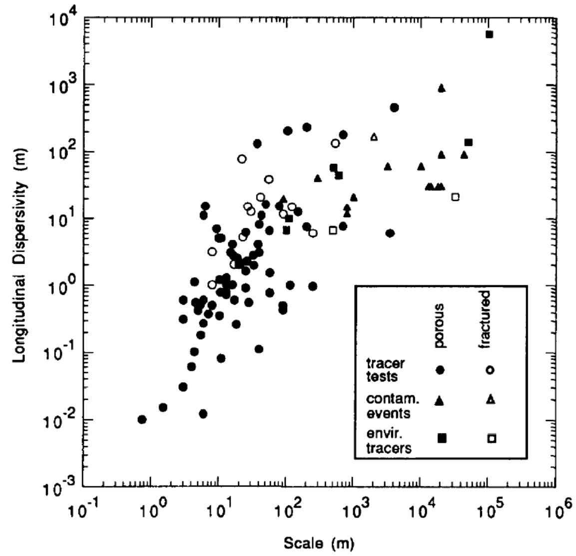
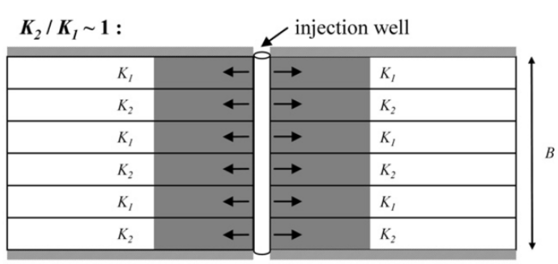


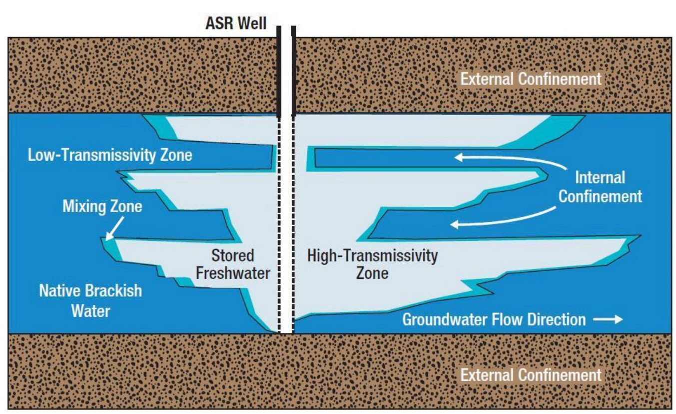
Figure 12: Longitudinal dispersivity versus scale of observation (Gelhar et al., 1992).

2.4.5 Heterogeneity Effects

When water is injected into an aquifer it spreads out, displacing ambient groundwater. Depending on the level of aquifer heterogeneity, the water spreading out will form a ‘bubble’ or ‘bottle brush’ formation (Lowry and Anderson, 2006). With the bottlebrush formation, freshwater passes more easily through the high transmissivity zones and forms the “bristles” of the brush (Lowry and Anderson, 2006). Figure 13a/b show the bubble versus bottle brush effect.



a)



b)

Figure 13: a) Bubble formation in a homogenous aquifer (Ward et al., 2008) and b) Bottlebrush formation in a heterogeneous aquifer (Maliva and Missimer, 2010).

The presence of large heterogeneities can have a significant effect on the system RE, especially where freshwater is injected into a brackish or saline aquifer. Water can migrate beyond the capture zone of the extraction well in high transmissivity zones, while water available for extraction can be lost to residual volume in low transmissivity areas, both actions causing a reduction in RE. Heterogeneity at a site can have dramatic impacts on the viability of the system and should be examined before pursuing a site for ASR operations. It is difficult, however, to determine the level of heterogeneity in an aquifer because of the high cost of subsurface investigations (Pavelic et al., 2006b). Tracer tests, thermal profiling and numerical modelling, as suggested by Pavelic et al. (2006b), can help to better characterise the aquifer and improve efficiency.

2.4.6 Clogging Effects

Clogging of injection wells is a significant challenge in the successful long-term operation of ASR/ASTR systems. Clogging can be physical and caused by the accumulation of suspended solids in the infiltration structure; biological and caused by the accretion of algae, bacterial flocs or other microorganisms on the infiltrating surface or surrounding porous media forming a biofilm; or chemical and caused by the precipitation of various chemicals onto the infiltrating surface or surrounding porous media (Bouwer, 2002). Clogging of recharge wells is a major concern even when dissolved salts in injection water are low, as is the case with recharged rainwater (Bouwer, 2002). Figure 14 is a photo of a clogged recharge well at one of the Bangladesh ASTR sites. In order to avoid serious clogging of wells and potentially permanent and/or expensive damage to equipment, regular backwashing of the recharge area should take place (Bouwer, 2002).

Figure 14: Clogged injection well.

Prior to implementing an ASTR scheme, thorough investigation must take place to ensure that introducing water from a foreign source to the aquifer does not cause pollution of the groundwater if the groundwater is already of a high quality (Dillon et al., 2009). Water intended to be injected into brackish aquifers, however, typically does not require as much pre-treatment as water injected into fresh aquifers because the ambient groundwater is already of a lower quality than drinking water (Dillon et al., 2009; Dillon et al., 2010)

Pavelic et al. (2006a) examined a 5-year, 250ML ASR trial where fresh, turbid urban stormwater water was injected into a brackish limestone aquifer to examine clogging and unclogging rates and the suitability of the site to be a source for irrigation water. Injection well clogging occurred frequently because of the high turbidity of the injection water and the chemical precipitation that occurred because of chemical differences between the background and injection water (Pavelic et al., 2006a). Sufficient filtration must take place to prevent clogging in the case that the receiving aquifer is fine-grained (Dillon et al., 2009). Airlifting after each injection-recovery cycle, where compressed air is forced through the well to remove clogging material, in combination with frequent monitoring is also recommended as it can ensure that system health is maintained and injection wells remain fully functional (Pavelic et al., 2006a).

Membrane filtration index (MFI), determined graphically from the slope of the time over volume of water passed through a membrane versus the volume of water passed through a membrane, is the metric used to classify injection water based on the likelihood of that injection water to cause physical clogging of the well material. The higher the MFI value, the more likely it is for clogging to occur (Pavelic et al., 2006a). Given varying source waters to be used at the Bangladesh sites, determining an average MFI criterion for the injection water at each site could prove useful for combating clogging problems. Additional treatment of source water might be necessary in the future.

2.5 Summary

This chapter reviewed the literature related to MAR technologies and provided background for the focus of this thesis topic. Several numerical models and field trials investigating ASR technology have been developed in the past; however, limited studies on ASTR technology, especially in the context of developing countries, have been conducted. General conclusions regarding ASTR systems can be made based on the modelling work involving ASR systems (e.g. RE can be reduced when the system is under the influence of a regional hydraulic gradient, variable density groundwater effects, high levels of dispersion or heterogeneity, and clogging) since the systems operate under similar governing physics, however, specific guidelines for ASTR development have not been formed.

Approximate system RE values for given hydrogeological settings and non-dimensional design parameters have be formed for ASR systems from numerous field work and numerical modelling studies and can be used to guide future site selection and design of ASR schemes (Ward et al., 2009). Conversely, ASTR has only been featured in one prominent numerical study which examined a large scale system (100+ ML injected per year) and was used to develop a site-specific well field design, and one field study of a large scale system investigating the effectiveness of ASTR at removing pathogens (Pavelic et al., 2004; Sidhu et al., 2010). Both systems were implemented in well developed countries with abundant physical and monetary resources. Non-dimensional design guidelines have not yet been developed so as to easily apply ASTR technology to different scenarios and approximations of expected system RE given site hydrogeological conditions have not been documented in the literature. Additionally, ASTR systems have not yet been investigated at the community-scale in a developing country context.

This study focuses on the development and application of a numerical model to simulate the feasibility of a small-scale (0.3 – 1.5 ML injected per year) ASTR operation in rural, coastal Bangladesh and to quantify the effect of site hydrogeological and engineering design parameters on system efficiency, such as aquifer hydraulic head changes caused by tidal action. Non-dimensional design parameters were developed to aid in future ASTR construction in other coastal communities.

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Chapter 3

3 NUMERICAL EVALUATION OF COMMUNITY-SCALE AQUIFER STORAGE, TRANSFER AND RECOVERY TECHNOLOGY

3.1 Introduction

There is an increasing need to develop disaster resilient, low-cost drinking water supply options, with many areas worldwide experiencing more frequent water shortages and natural disasters, combined with rapid population growth (Maliva and Missimer, 2010). Managed aquifer recharge (MAR) is a technology capable of addressing water supply issues for communities with seasonal water scarcity. In MAR, water from the surface or from another water source is injected into the ground via engineered systems such as wells, trenches, infiltration ponds or percolation tanks (Bouwer, 2002; Dillon et al., 2009).

Aquifer storage, transfer and recovery (ASTR) is a type of MAR where surface water is injected directly into the aquifer, creating a water store that can later be used as a water source (Maliva and Missimer, 2010). Water is injected into the aquifer through an injection well and extracted some distance away via an extraction well (Maliva and Missimer, 2010). Injected water is naturally filtered by the aquifer media as it is transported from the injection well to extraction well, which removes bacteria and other pathogens (Page et al., 2010). ASTR has previously been implemented in both fresh and brackish aquifers with reasonable (~50%) recovery efficiencies (RE, the ratio of the volume of water which can be extracted at a suitable standard to the volume of water injected) (Ward et al., 2009).

This study focuses on evaluating the feasibility of ASTR technology for coastal communities in south-western Bangladesh. Communities in the low-lying Khulna-Satkhira-Bagerhat coastal region (Figure 15a) are experiencing increasingly severe seasonal water scarcity (Hasan, 2012). In this region, the groundwater is naturally brackish and traditional drinking water sources (e.g., surface ponds) are frequently contaminated from episodic weather events such as cyclones and storm surges (Karim and Mimura, 2008). ASTR is currently being tested by UNICEF-Bangladesh, University of Dhaka and Acacia Water (Netherlands) as a potential cost-effective, disaster-resilient water supply alternative for the coastal communities. Though the use of this technology does result in some losses of the injection water caused by mixing, the benefits of securing water supplies against the frequent natural disasters in the coastal areas during a time when freshwater is plentiful outweigh the drawbacks of these water losses. Since 2011, twenty test sites have been established where freshwater, collected via rooftop rainwater harvesting and surface ponds, is injected into shallow aquifers during the monsoon season and stored for extraction throughout the year (Hasan, 2012).

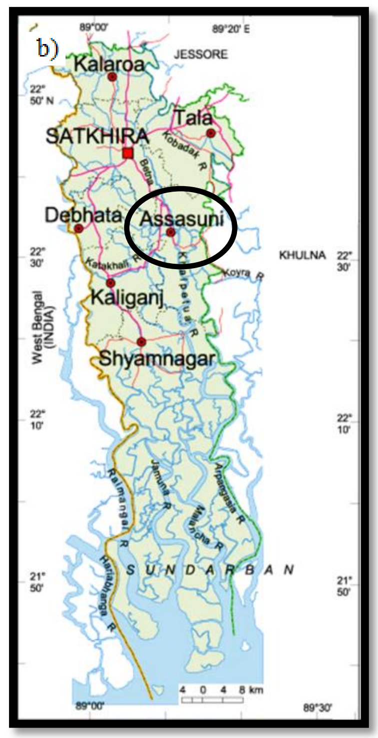

Figure 15: a) Map of Bangladesh (study region highlighted) (U.S. Department of State, 2013) and b) Satkhira district map showing Assasuni (PATH Foundation Bangladesh, 2007).

A numerical groundwater model of the ASTR system at the Assasuni site, located in the Satkhira district, was first developed to investigate the long-term feasibility of the technology based on the current system design and hydrogeological conditions. Sensitivity analyses were then performed to improve future field site selection and the engineered system design. Following this, a generic ASTR model was created to analyze the system in a non-dimensional framework and to establish design criteria that can be applied for sites in other coastal regions worldwide that face similar seasonal water scarcity issues.

3.2 Site Description

The ASTR system at the Assasuni site (Figure 15b, N +22° 32' 50.98", E +89° 10' 40.84", Appendix A) has been operational since June 2011 and the hydrogeological and operating conditions at this site are well characterized (Hasan, 2012). The aquifer into which the ASTR system injects water is naturally brackish with a total dissolved solids (TDS) concentration of ~4.9 g/L. The average concentration of the injection water sourced from rainwater harvesting and a nearby pond is ~0.6 g/L. The TDS concentration of the extracted water should not exceed the Bangladesh Drinking Water Standard (BDWS) of 1 g/L (Rahman and Bhattacharya, 2006). TDS concentrations and salt concentration are synonymous for the purposes of this study as the magnitude of the concentration of other TDS (e.g., nitrate, phosphorus, iron; size < 2 microns) are low relative to salt (Hasan, 2012; U.S. EPA, 2012).

The ASTR system includes four injection wells of varying diameter (0.3048m or 0.5588m): two wells are screened over the entire depth of the alluvial, sand aquifer (z = -13m to z = -24m) and two wells are screened from z = -13m to z = -14m. (Figure 16) (Hasan, 2012). Note: the vertical direction is denoted by z. There is a single extraction well offset from the injection well area by ~0.5m that is screened in the upper layers of the aquifer (z = -13m to z = -17m). A 13m thick clay aquitard ($K_x = K_y = 1 \times 10^{-3}$ m/d, $K_z = 1 \times 10^{-4}$ m/d, n_e = 0.25) overlies the sandy aquifer ($K_x = K_y$ = 0.2 m/d, K_z = 0.1 m/d, n_e = 0.25) and this prevents the infiltration of surface water into the aquifer (Hasan, 2012) (site borehole log and injection well diagram are included in Appendix A). A tidally-influenced river, which is depressed from the land surface level and therefore hydraulically connected to the aquifer, runs approximately 200m to the northwest, resulting in diurnal water table fluctuations (approximately 0.15-0.3 m) at the site. The regional hydraulic gradient influencing the system is negligible as per the work conducted by Michael and Voss (2009).

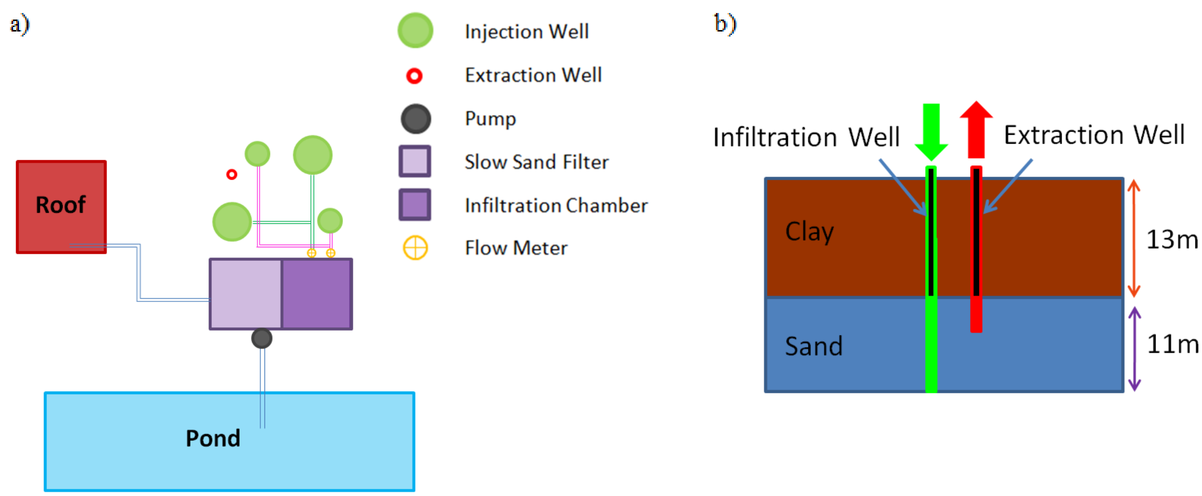

Figure 16: Assasuni ASTR site layout (not-to-scale) a) simplified plan view, b) simplified cross-sectional view.

Rainwater is collected on the rooftop and gravity fed to the slow sand filter while water collected from the pond is pumped to the slow sand filter. The sand filter removes particulate matter present in the source water. The filtered water then enters an infiltration tank from which it is distributed via pipeline to the injection wells. The injection wells are gravel packed and serve as a high permeability zone for the injection water to bypass the thick clay layer and enter the aquifer. At the site a locally-trained site operator records daily injection volumes, extraction volumes, local weather conditions and also maintains the head levels in each of the injection wells at ~1 m above the ground surface. As the ambient groundwater level at the Assasuni site is ~0.58 m below ground surface, a total injection head of 1.58 m is therefore maintained in the injection wells.

3.3 Numerical Model

A three-dimensional numerical model was developed in the variable-density groundwater flow and transport code SEAWAT-2005 to simulate the ASTR system at the Assasuni site. The governing equations solved by SEAWAT-2005 are provided in Appendix B. The model was first calibrated for the Assasuni site and following this the model was applied to investigate the influence of a range of hydrogeological and engineering design parameters that were shown from the literature to have a large impact on ASTR system efficiency.

The performance of the ASTR system was assessed based on the following criteria: (i) the TDS of the extracted water must be at or below the BDWS of 1 g/L (for this study TDS is an integrated measure of water quality); (ii) RE should be maximized (RE is the ratio of volume of water extracted below 1 g/L to the volume of water injected); and (iii) a residence time of 2.5 days or more in the aquifer must be achieved to ensure the removal or inactivation of microbial contaminants in the aquifer (Rahman and Bhattacharya, 2006; Sidhu et al., 2010; Ward et al., 2009; World Health Organisation, 2003).

3.3.1 Model Set-up

A numerical model of the Assasuni site was set-up to best represent the site conditions and well operating schedules (Figure 17). The model domain was 800m x 800m x 24m (*x*, *y* and *z*-directions respectively). A low hydraulic conductivity zone representing the clay aquitard (13m deep) overlies a higher hydraulic conductivity zone representing the sand aquifer (11m deep). The parameters used for this model (referred to as the base model) are provided in Table 2.

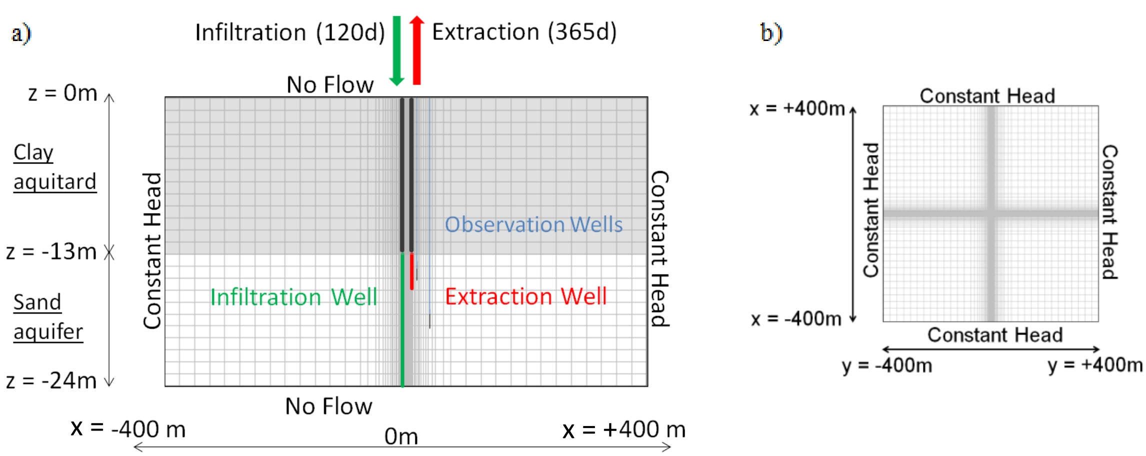

Figure 17: Model set-up (not-to-scale) a) cross-section (note: one injection well shown for simplicity) and b) plan view. The flow boundary conditions and uneven grid discretization are shown.

Table 2: Base model parameters.

Parameter	Value
Ambient Groundwater Head	-0.58m
Injection Head (*H*)	1.58m
Extraction Rate	1.029 m^3/d
Diameter of Injection Wells	Two at ϕ = 0.3048 m, Two at ϕ = 0.5588 m
Number of Injection Wells	4
Number of Extraction Wells	1
Distance From Injection Wells to Extraction Well	0.5 m
Aquifer Thickness (*B*)	11 m
Injection Well Screen Length	2 screened for 11 m (z = -13m to z = -24m) 2 screened for 1 m (z = -13m to z = -14m)
Extraction Well Screen Length	3 m (z = -13m to z = -16m)
Longitudinal Dispersivity (α_L)	2.5 m
Longitudinal/Transverse Dispersivity (α_L/α_T)	0.1
Aquitard Hydraulic Conductivity (*K*)	$K_x = K_y = 1 \times 10^{-3}$ m/d (longitudinal) $K_z = 1 \times 10^{-4}$ m/d (vertical)
Aquifer Hydraulic Conductivity (*K*)	$K_x = K_y = 0.2$ m/d (longitudinal) $K_z = 0.1$ m/d (vertical)
Regional Hydraulic Gradient (*i*)	0
Effective Porosity (n_e)	0.25
Ambient TDS Concentration	4.95 g/L
Injection Water TDS Concentration	0.587 g/L

Constant heads are applied to the outer model boundaries to simulate the regional hydraulic gradient (negligible gradient for base case) and to prevent artificial water table mounding around the ASTR system. The lower and upper model boundaries are no flow boundaries thus assuming an impermeable aquifer basement and no recharge into the aquifer. The injection well is simulated using a general head boundary (GHB) specified head cell. The model has 86 rows, 86 columns and 24 layers and is more highly discretized at the centre to yield a grid Peclet number $\left(\frac{v_x \Delta_x}{D_x}\right)$ is less than 2 to ensure model stability (Karniadakis and Kirby, 2003). Grid independence and model domain size tests were performed to ensure the solution was converged (see Appendix C). The base model simulating the Assasuni ASTR system was run for a total simulation time of five years. Within each year, injection occurred for 120 days and extraction occurred for 365 days to simulate the existing injection-recovery schedule for the site.

3.3.2 Model Calibration

The base model of the Assasuni system was calibrated by varying the aquifer hydraulic conductivity and longitudinal dispersivity and comparing the simulated and measured TDS concentrations and injection rates. Aquifer hydraulic conductivity values were varied until there was a close match between the observed field injection rate (4.57 m^3/d) and the simulated injection rate (4.85 m^3/d). From this the aquifer longitudinal dispersivity was varied to yield a close match between site TDS concentration data and simulated concentration data. Figure 18 shows reasonable comparison between the measured concentrations and those predicted by the calibrated model at the extraction well during the first injection period (day 0 to day 120) and at an observation well 5m away from the injection well area at the beginning (day 385 to day 400) of the second injection period (total injection period from day 365 to day 505). Discrepancy between the site and model data shown in Figure 18a is caused by inconsistent (non-constant) injection heads at the site due to poor site monitoring by local, minimally-trained personnel. Nevertheless, the consistency between the simulated and measured values for the TDS concentration and injection rate indicated that the calibrated model performance was acceptable.

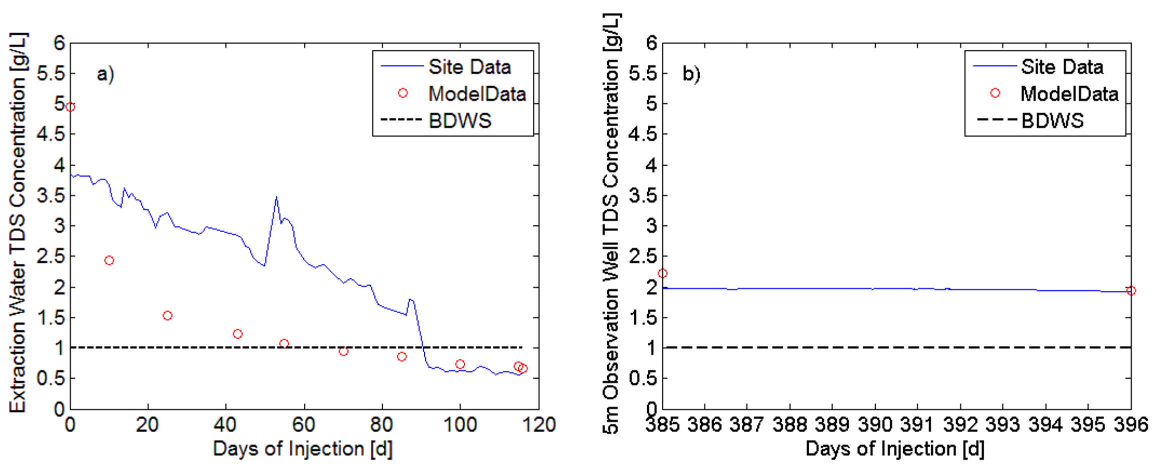

Figure 18: Comparison between observed TDS concentrations and calibrated model results a) at extraction well (*x* = +1.5m) during first injection period (day 0 to day 117) and b) at 5m observation well (*x* = +5m) during beginning (day 385 to day 396) of second injection period. Fluctuations in measured TDS concentrations are due to variability in the site injection head. The Bangladesh Drinking Water Standard (BDWS) (1 g/L) is indicated by the horizontal dashed black line.

The calibrated hydraulic conductivity and longitudinal dispersivity were K_x = 0.2 m/d and K_z = 0.1 m/d, and α_L = 2.5m respectively. Values for the aquifer hydraulic conductivity were independently obtained through site slug tests and compared to these calibrated values. From the slug tests, K_x was determined to be approximately 1.8 m/d. This value matches well with the calibrated value of 0.2 m/d. For the model scale and a sandy aquifer, a longitudinal dispersivity of 2.5 is quite reasonable (Gelhar et al., 1992; Schwartz and Zhang, 2003).

3.4 Model Results

3.4.1 Performance of ASTR system at Assasuni site

Five annual injection-recovery cycles were simulated with the calibrated model of the Assasuni site (referred to as the base model). As expected, injected water entered the aquifer via the injection wells and formed a plume of fresher groundwater that was available for extraction (Figure 19). A brackish-freshwater transition (mixing) zone was present around the plume. Low RE values are typically experienced in the start-up phase (1-2 years) of an ASTR operation as the freshwater plume becomes established (Ward et al., 2009). This occurred for the base model where after the first year (Figure 19a), only a small freshwater plume had developed and extraction of some ambient groundwater took place, raising extraction TDS concentrations to 3.35 g/L (Figure 20). Over time, however, the freshwater plume diameter increased (Figure 19b). The fluctuations in plume diameter and TDS concentrations are caused by successive injection-recovery cycles. While extraction TDS concentrations could not be maintained below the BDWS, towards the end of the extraction-only period the maximum extraction TDS concentrations did reduce to 1.35 g/L by the end of year 5 (Figure 20a). The site design with the extraction well offset from the injection wells (rather than centred in the middle of the injection wells) and the extraction well only screened in the upper 3 metres of the aquifer caused a decrease in the plume size at the top of the aquifer and ambient groundwater was more readily drawn into the extraction well (Figure 19b). The system was able to achieve a 4.9 day aquifer retention time so it is predicted that the presence of bacteria in the extraction water will not be a concern.

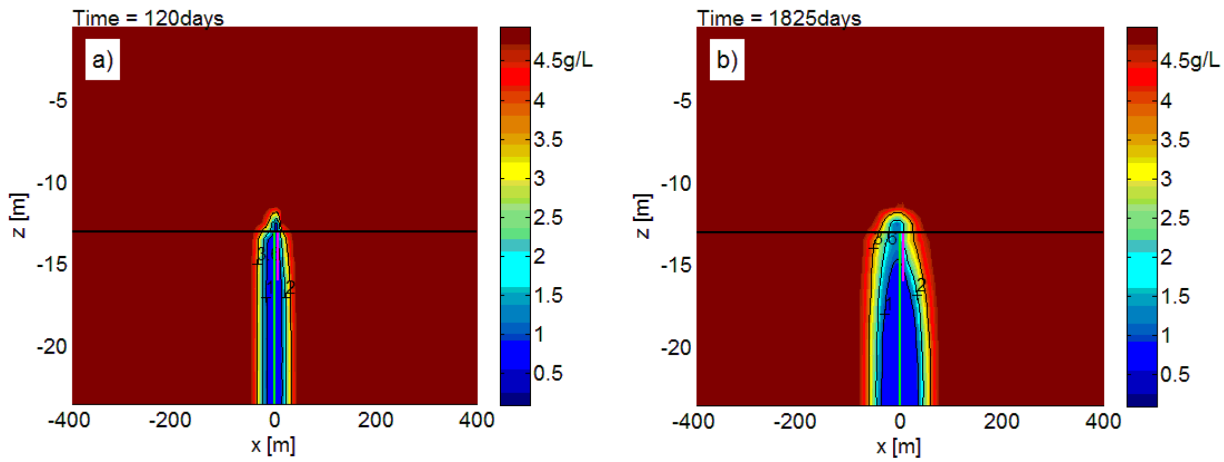

Figure 19: Cross-sectional view of simulated TDS distribution for the base model a) after 1 year (365 d) and b) after 5 years (1825 d). The black horizontal line denotes the top of the aquifer layer.

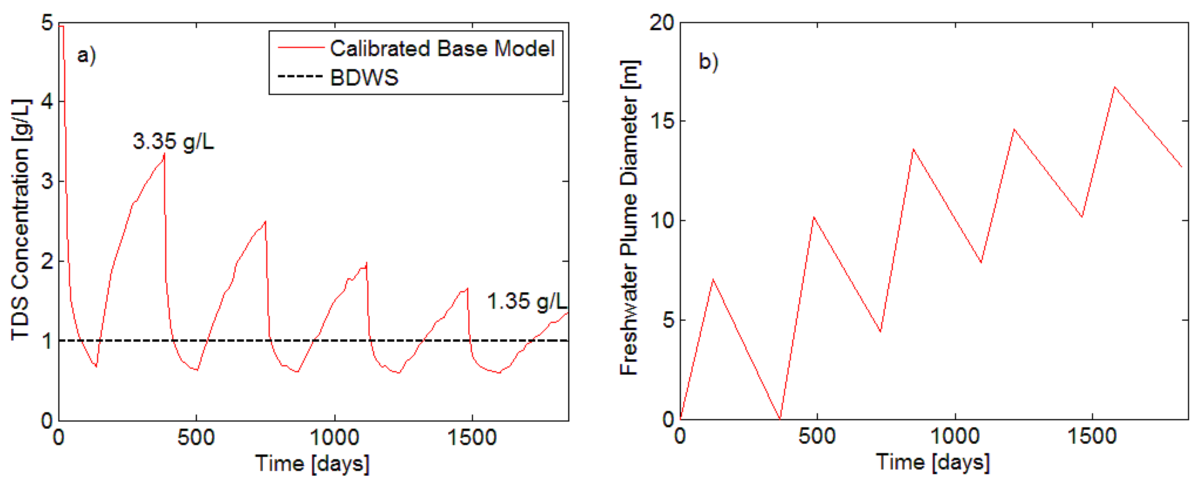

Figure 20: a) TDS concentrations at extraction well, and b) calculated freshwater plume diameter for the base model from day 0 to day 1825. Plume diameter is taken as the width of the plume contained within the 1g/L contour level.

Spatial moments analysis was conducted to quantify the time-varying characteristics of the injected freshwater plume including the mass of freshwater in the system (zeroth moment), the location of the plume (first moment) and the spread of the plume (second moment) (Barry and Sposito, 1990). The equations used for the spatial moments analysis are presented in Appendix D. Figure 21 shows the calculated spatial moments for the base model. As expected, the mass of freshwater in the aquifer increased with time, with the annual mass fluctuations caused by the seasonal injection and recovery cycles (Figure 21a). The plume remained relatively centred (centre of model at x = 0, z = -19), with a slight progressive shift leftward as the injection area is slightly offset leftwards of the model centre (Figure 21b,c). The spread of the freshwater plume (Figure 21d,e), quantified in the *x*-direction by σ_{xx}^2 and in the *z*-direction by σ_{zz}^2, was low with σ_{xx}^2 slowly increasing in response to the injection-recovery cycles.

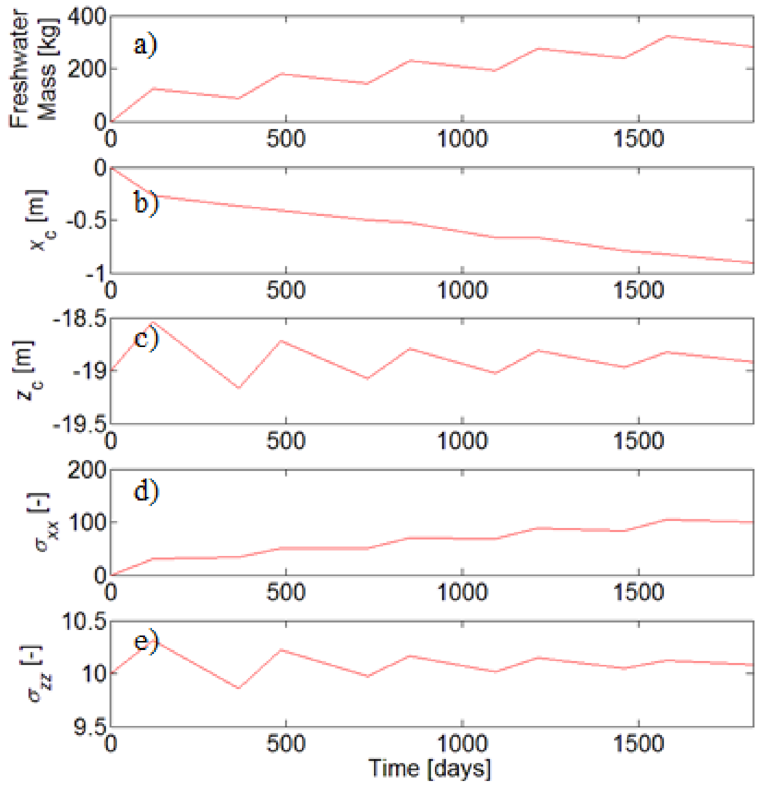

Figure 21: Calculated spatial moments for base model: a) mass of freshwater in the aquifer, b) *x*-coordinate of the centroid of the freshwater plume, c) *z*-coordinate of the centroid of the freshwater plume, d) extent of spreading/mixing in the *x*-direction, and e) extent of spreading/mixing in the *z*-direction.

3.4.2 Sensitivity Analyses

Sensitivity analyses were conducted to examine the influence of various hydrogeological and engineering design parameters. The selection of which sensitivity analyses to perform was determined based on the available input parameters to the model and from informed predictions from the review of the literature of which parameters would have a large influence on the system performance. Each sensitivity parameter examined was changed for the base model condition while all other parameters pertinent to the base model were maintained at the base model value. A summary of simulations conducted and results for the scenarios which had the largest impact on system performance are provided in Table 3 with additional sensitivity analyses and detailed results provided in Appendix E. Similar to the base model, each simulation was run for a total of five years (excluding the model including tidal head fluctuations which was run for only 1 year to reduce computational time) with injection occurring for the first 120 days of each year and extraction occurring over the full 365 days. Values provided in Table 3 are for the end of the simulation (1825 d). Injection rates reported in Table 3 are the steady-state rates reported at the end of the injection period. Plume radii were calculated at z = -19m (mid-aquifer depth) for concentrations contained within the 1 g/L contour level.

Table 3: Summary of simulations conducted and results after 5 years (1825 d). The parameter used for the base model is bolded for each sensitivity analysis case.

Sensitivity analysis	Parameter Value	Injection Rate [m^3/d]	Plume Radius [m]	Freshwater Mass [kg]	Plume Centroid (x_c,z_c)	Mixing σ_{xx}^2	σ_{yy}^2
Aquifer Hydraulic Conductivity (*K*)	K_x = 0.1 m/d K_z = 0.05 m/d	2.9	0	143	(-0.5, -19.3)	65	10
	K_x = 0.2 m/d K_z = 0.1 m/d	4.9	6	283	(-0.9, -18.9)	100	10
	K_x = 1.0 m/d K_z = 0.5 m/d	20.6	24	1900	(-0.4, -18.1)	676	11
	K_x = 5.0 m/d K_z = 2.5 m/d	99.1	56	15800	(-0.2, -18)	4420	11
	K_x = 10.0 m/d K_z = 5.0 m/d	197.3	388	36600	(0, -18)	7728	12
Aquifer Longitudinal Dispersivity (α_L)	1.0m	4.9	9	264	(-0.9, -19)	77	9
	2.5m	4.9	6	283	(-0.9, -18.9)	100	10
	5.0m	4.9	0	311	(-0.8, -18.7)	165	11
	10.0m	4.9	0	353	(-0.7, -18.4)	291	12
Aquifer Thickness (*B*)	5 m	2.7	0	58	(-0.9, -15.2)	73	3
	11 m	4.9	6	283	(-0.9, -18.9)	100	10
	20 m	6.3	19	420	(-0.9, -21)	87	22

	30 m	6.9	21	474	(-1, -21.2)	89	23
	40 m	7.1	23	535	(-1, -23.4)	86	79
Hydraulic Gradient (*i*), flowing from y = -400 m boundary to y = +400 m boundary	**0**	4.9	6	283	(-0.9, -18.9)	100	10
	0.0005	4.7	5	282	(0.4, -18.9)	107	10
	0.0015	4.8	1	302	(1.4, -18.8)	142	11
	0.0025	4.8	1	307	(2.9, -18.8)	141	11
	0.005	4.7	2	348	(7.3, -18.8)	151	11
Injection Head (*H*)	**1.58 m**	4.9	6	283	(-0.9, -18.9)	100	10
	2.58 m	6.92	12	468	(-1, -18.7)	143	10
	3.58 m	9.0	16	662	(-0.8, -18.5)	209	10
	4.58 m	11.1	18	868	(-0.8, -18.4)	293	11
Extraction Rate (Q_{ext})	0.5 m^3/d	4.4	10	367	(-0.7, -18.6)	116	10
	1.029 m^3/d	4.9	6	283	(-0.9, -18.9)	100	10
	2.0 m^3/d	5.6	0	196	(-1, -19.3)	96	10
	3.0 m^3/d	6.4	0	143	(-0.8, -19.5)	90	10
	5.0 m^3/d	8.1	0	84	(-0.7, -19.8)	77	9
	8.0 m^3/d	10.5	0	44	(-0.3, -20.1)	59	9
Injection Well Number/Spacing (ϕ= 0.3048m)	2	4.3	4	237	(-1, -19)	100	10
	4, 0.6m spacing	4.7	5	266	(-1.1, -18.9)	107	10
	4, 1.8m spacing	4.9	6	312	(-1.2, -18.9)	115	10
Tidal Effects (1 year results)	**$\Delta H_{aquifer}$ = 0m**	4.9	0	87	(-0.4, -19.2)	33	10
	$\Delta H_{aquifer}$ = 0.3m	7.5	0	142	(-0.4, -18.7)	60	11

3.4.2.1 Hydrogeological Parameters

The hydrogeological parameters tested had a large impact on the RE of the ASTR system. As the ASTR systems currently implemented in south-western Bangladesh are designed to achieve a constant injection head (rather than constant injection rate), higher aquifer hydraulic conductivity, when present at a site, led to higher injection rates and thus greater freshening of the aquifer (e.g., Q_{inj} = 2.9 m^3/d for K_x = 0.1 m/d c.f. Q_{inj} = 197.3 m^3/d for K_x = 10m/d; Table 3). A 2x increase in K_x, going from K_x = 0.1 m/d to 0.2 m/d, resulted in a 1.7x increase in injection rate, while a 100x increase in K_x, going from K_x = 0.1 m/d to 10 m/d, resulted in a 68x increase in injection rate. The size of the freshwater plume at five years increased from 0m for K_x = 0.1 m/d to 388m for K_x = 10m/d. As a result, the RE increased and the extraction concentrations were able to meet and go below the BDWS standard when the aquifer had a higher *K* (Figure 23a). Simulations indicate that if an ASTR system is to be installed in an aquifer of low *K*, modifications to the system design (e.g. higher injection head, additional injection wells; see Section 3.4.2.2) would be required to improve the system performance. Higher *K* values increased velocities in the *x*-direction which served to shift the centroid of the freshwater plume and increase the width of the brackish-freshwater mixing zone (σ_{xx}^2 = 65 and 7728 after 5 years for simulations with K_x = 0.1m/d and K_x = 10m/d, respectively). The larger freshwater plume created in the aquifer with higher *K* was, however, able to overcome the negative effects of greater mixing on the extraction water TDS concentrations (Figure 232a).

The aquifer thickness (*B*) was varied from 5 – 40 m with the injection well screen length kept constant at 11 m, spanning from *z* = -13m to *z* = -24m. The exception was for the simulation with *B* = 5 m where the screen length was reduced to 5 m (injection well screen spans from *z* = -13m to *z* = -18m). Similar to the effect of varying the aquifer hydraulic conductivity, the injection rates increased as the aquifer thickness increased (Q_{inj} = 2.7 m^3/d for *B* = 5 m c.f. Q_{inj} = 7.08 m^3/d for *B* = 40 m). A 1.8x increase in the aquifer thickness, going from *B* = 11m to *B* = 20 m, resulted in a 2.3x increase in the injection rate, while a 3.6x increase in *B*, going from *B* = 11m to *B* = 40 m, resulted in a 2.6x increase in the injection rate. Increased injection rates led to reduced TDS concentrations at the extraction well and improved RE (Figure 232b). As *B* increased, the plume became more ovular and spread out in the *z*-direction as evidenced by the downward shift in the plume's centroid and the increase in σ_{zz}^2 (Table 3). While the plume radius increased as *B* increased, the lateral spreading (σ_{xx}^2) was relatively constant (Table 3). Additional simulations performed to examine the impact of the extraction well screen length and screening depth showed that the RE may be maximized by extending the extraction well screen over the entire aquifer depth if the aquifer can be fully screened in given site conditions (see Appendix E for more information). If the aquifer cannot be fully screened because of limitations in the drilling process (wells are typically drilled by hand in Bangladesh for economic reasons meaning that drilling to depths greater than ~25m is often not feasible), the extraction well screen depth should be located towards the bottom of the injection well to take into account the downward shift in the plume (Ahmed, 2012).

Aquifer dispersivity is difficult to measure in the field but has a significant impact on the TDS concentrations at the extraction well as larger dispersivities increase mixing between the ambient groundwater and injected water (Figure 23c). Sites with large aquifer dispersivity may be unusable if the brackish-freshwater mixing is too great, as in the case of α_L = 5.0 m and 10.0 m for the conditions simulated. Increasing α_L from α_L = 1.0 m to 2.5 m caused a 1.3x increase in the amount of mixing/spreading in the *x*-direction, while increasing α_L from α_L = 1.0 m to 10.0 m resulted in a 3.8x increase in the amount of mixing/spreading. It can be seen in Table 3 that for α_L = 5.0 m or 10.0 m, no distinct freshwater plume formed even after five years of injection because the mixing in the system was too high for the concentrations of the extracted water to decrease below 1 g/L. The σ_{xx}^2 value increased as α_L increased from 1.0 to 10.0, leading to a decrease in system RE. The σ_{xx}^2 value for the simulation with α_L = 10.0 m was low compared to the σ_{xx}^2 values calculated for the simulations with high aquifer hydraulic conductivity, however, the system with α_L = 10.0 m does not have an accompanying increase in injection flow rate to overcome the mixing effects as the *K* models do; therefore these systems perform poorly in comparison. As expected, α_L did not cause a change the injection rates and thus the freshwater mass in the system was fairly constant between models.

Although the regional hydraulic gradient at the Assasuni site is negligible, regional hydraulic gradients often affect the efficiency of ASTR systems (Ward et al, 2009). Simulations showed that a high regional hydraulic gradient will cause the freshwater plume to shift downstream, away from the extraction well, resulting in a slight increase in TDS concentration of the extracted water (Figure 23d). It is expected that if the hydraulic gradient were increased further there would be a more noticeable effect on the extracted water TDS concentration. For the model with i = 0.0005, the freshwater plume centroid migrated from (-0.9,-18.9) to (0.4,-18.9), a shift of 1.3 m; while for the model with i = 0.005, the freshwater plume centroid migrated from (-0.9,-18.9) to (7.3,-18.8), a shift of 8.2 m. This movement also led to greater σ_{xx}^2 values, whereas σ_{zz}^2 was relatively constant between simulations. The radius of the freshwater plume decreased as the hydraulic gradient increased due to the enhanced brackish-freshwater mixing induced by higher groundwater flow velocities.

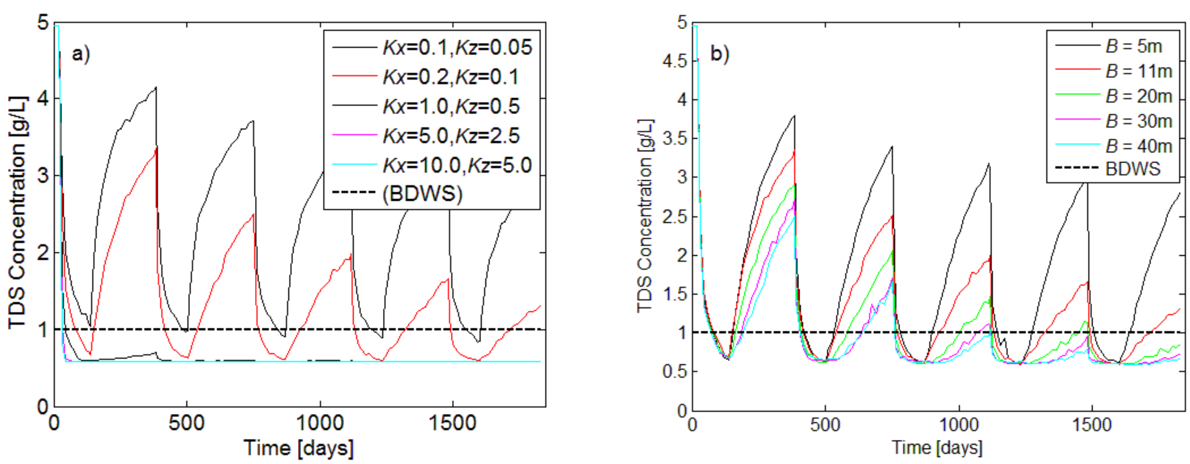

Figure 22: TDS concentration of extracted water over 5 year simulation period for simulations with varying a) aquifer hydraulic conductivity (K_x and K_z are the hydraulic conductivity in the longitudinal/transverse and vertical planes, respectively), and b) aquifer depth (B).

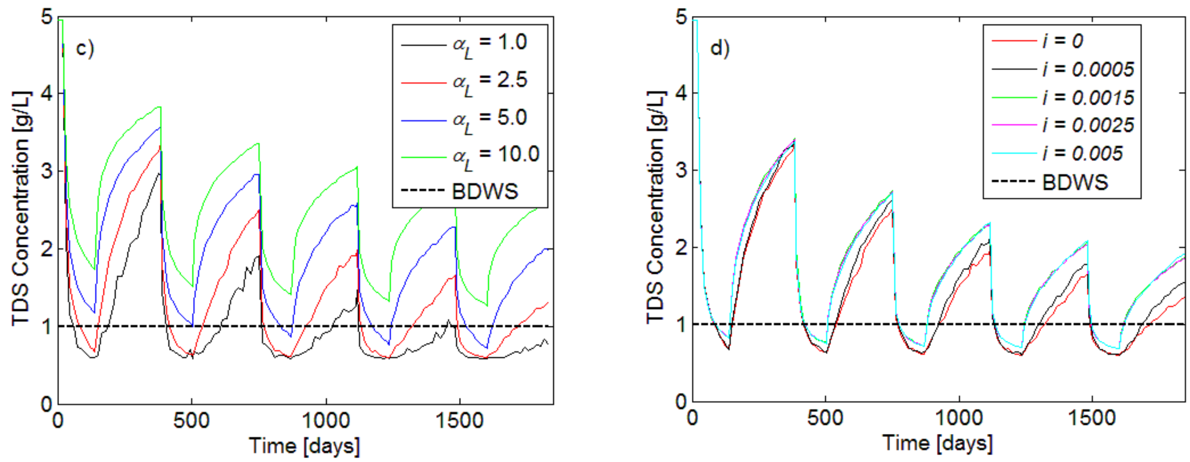

Figure 23: TDS concentration of extracted water over 5 year simulation period for simulations with varying c) aquifer longitudinal dispersivity (α_L), and d) regional hydraulic gradient (*i*).

3.4.2.2 Engineering Design Parameters

Simulations conducted with varying injection hydraulic head (ranging from H = 1.58 m to H = 4.58 m) showed that, as expected, increased injection head leads to higher injection rates and a greater mass of freshwater in the aquifer (2830 kg and 8680 kg of freshwater mass in the aquifer after 5 years for H = 1.58 m and H = 4.58 m, respectively; Table 3). For a 1.6x increase in injection head, going from H = 1.58 m to 2.58 m, a 1.4x increase in injection rate resulted, while for a 2.9x increase in injection head, going from H = 1.58 m to 4.58 m, a 2.3x increase in injection rate resulted. This in turn resulted in lower TDS concentrations in the extracted water (Figure 24a). While the location of the plume centroid was relatively constant for the simulations with varying injection head, the mixing in the *x*-direction (σ_{xx}^2) increased due to the higher groundwater velocities. As site characteristics allow, injection heads should be kept as high as possible in order to improve the RE of the system. This could be achieved by hydraulically connecting the injection wells with water stored in a raised concrete injection tank, for example.

Increasing the water extraction rate caused ambient brackish water to be more readily drawn into the extraction well (Figure 24b). This was partly due to the spatial well layout currently used where the extraction well is offset from the injection wells (Figure 17a). Greater extraction rates did increase drawdown in the aquifer leading to increased injection rates (Table 3); however, these higher injection rates were not able to overcome the adverse effects of the higher extraction rates. As Q_{ext} increased from 0.5 m^3/d to 5 m^3/d, the Q_{net} decreased from 0.98 m^3/d to 0.77 m^3/d therefore decreasing the mass of freshwater present in the aquifer and the diameter of the freshwater plume. A 2.1x increase in the extraction rate, corresponding to increasing from Q_{ext} = 0.5 m^3/d to Q_{ext} = 1.029 m^3/d resulted in a 1.3x decrease in the mass of freshwater in the aquifer, while a 16x increase in extraction rate, going from Q_{ext} = 0.5 m^3/d to Q_{ext} = 8 m^3/d caused an 8.3x reduction in the mass of freshwater in the aquifer. The location of the plume was not noticeably affected by increased extraction rates but σ_{xx}^2 decreased as the extraction rate increased (Table 3). This was due to limited outward movement of the injected freshwater (injected water was rapidly removed from the aquifer via the extraction well rather than spreading out into the aquifer).

The base model has four injection wells: two wells with ϕ = 0.3048m and two with ϕ = 0.5588m. To evaluate the influence of the number of injection wells on the system performance, simulations with varying numbers of injection wells (each well had ϕ = 0.3048 m; models with varying well diameters were also tested and minimal effect on RE was observed, see Appendix E). An injection well spacing of ~0.6 m for two wells and four wells, in addition to ~1.8m for four wells was examined. The injection wells were placed in a square configuration with the extraction well offset. A well spacing of ~0.6m was simulated so as to create an aquifer retention time of 5 days (doubling the required 2.5 day retention time to create a factor of safety for the system) and then this well spacing was further increased to examine the effect of well spacing. The location of the extraction well was located ~0.6m to the right of the injection site for all cases. Figure 24c shows that the changing number of wells did impact the efficiency of the system as did the spacing between the injection wells. A 2x increase in the number of injection wells with 0.6m spacing resulted in a 1.09x increase in injection rate, while 2x increase in the number injection wells with 1.8m spacing (3x greater spacing) resulted in a 1.13x increase in injection rate. Given that the ASTR systems currently installed are designed to achieve a constant injection head, closely spaced wells result in greater localized mounding of the hydraulic head in the aquifer and this reduces the injection rate. This result implies that either increasing the number of injection wells or increasing the spacing between them can be used to positively impact the system efficiency and thus the extraction water TDS concentrations.

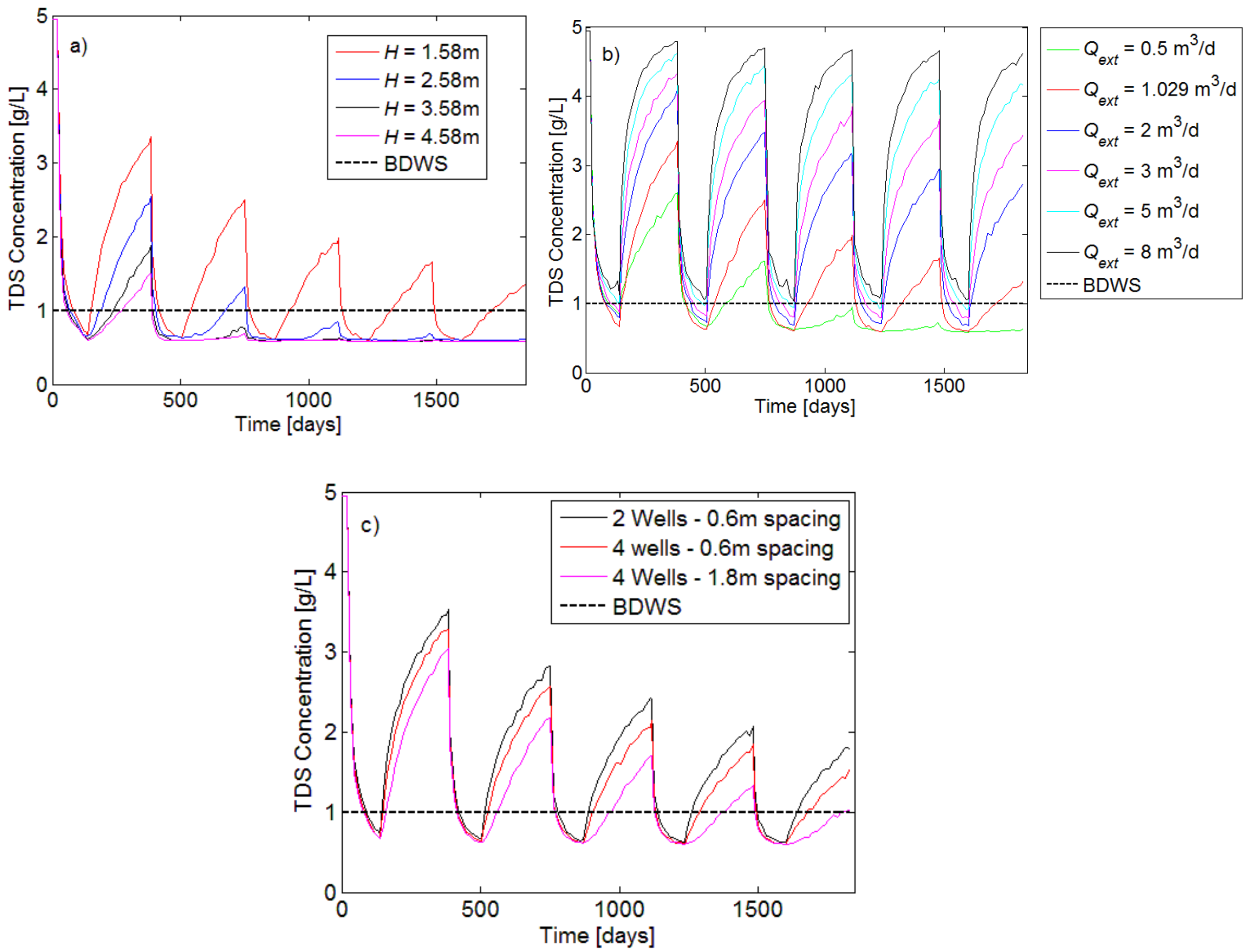

Figure 24: TDS concentration of extracted water over 5 year simulation period for simulations with varying a) injection head (*H*), b) extraction rate (Q_{ext}), and c) number of extraction wells (grid discretization in the well injection area is 0.3048m and, thus, the spacing of the injection wells is based on multiples of this grid discretization, Note: spacing values in the figure legend have been rounded).

3.4.2.3 Tidal Effects and Inundation Events

The Assasuni site experiences diurnal hydraulic head fluctuations ($\Delta H_{aquifer} \cong 0.3$m) as it is located ~200m to the north-west of a tidally-influenced river. The impact of the tideinduced head fluctuations on the ASTR system performance was simulated and it was found that the diurnal fluctuations had limited effect on the system RE although the extent of mixing in the *x*-direction did increase due to enhanced groundwater velocities (see Table 3, Appendix E).

The resiliency of the ASTR system to an overland saltwater inundation (flooding), as may occur during a storm surge or cyclonic event, was also examined. Simulations of a 5 day long saltwater inundation event were conducted where the inundation water had a TDS concentration of 20 g/L. Results suggest that for the conditions simulated an inundation event may lead to an increase in the TDS concentration at the extraction well above the BDWS (Figure 25). If the event occurs during the injection period the system may be able to rapidly recover, however, inundation events during the extraction phase may be more catastrophic causing the system to be unusable until the subsequent injection period. According to Karim and Mimura (2008), storm surges arising from cyclonic events typically occur from April to May in Bangladesh. This would correspond with an injection period and therefore the ASTR system would recover in the event that flooding breached the top of the raised injection structures. A storm surge may occur in the post-monsoon season (October-November), and in this case the ASTR system may not be a viable water source until the following monsoon season when injection would recommence (Karim and Mimura, 2008).

Given that the current system design incorporates injection wells that are only one meter higher than the land surface, the potential for seawater to enter the wells during a flooding event is much higher than if the design were modified so that the injection wells were not open to the free surface at the top of the injection well but rather hydraulically connected to the holding tank. This design would not only improve the ability of the system to ensure potable drinking water is injected even during a flooding event but would also increase the injection head for the system leading to higher injection rates into the aquifer and greater aquifer freshening.

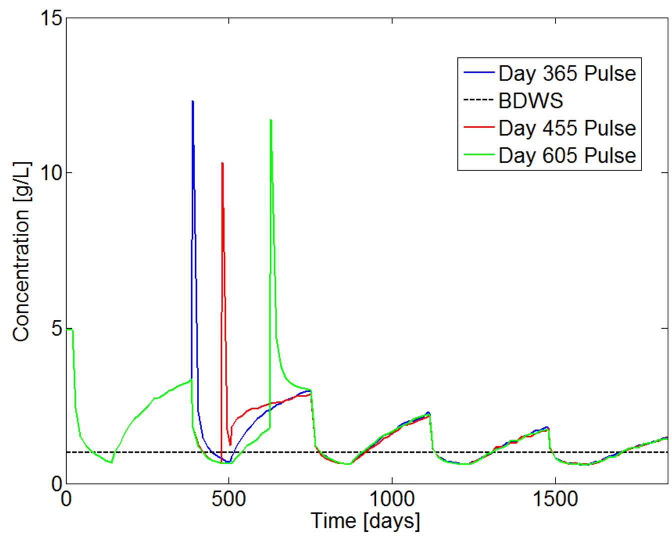

Figure 25: Impact of a five-day high concentration (20 g/L) pulse occurring at varying times in the injection-recovery cycle on extraction water TDS concentration.

3.4.3 Non-dimensional ASTR Design Guidelines

The numerical evaluation of the ASTR technology presented above focuses on modifications to the existing system design currently used in south-western Bangladesh. Here a more generic ASTR system design is adopted to better explore and provide design recommendations for future systems that may be implemented in a range of coastal settings. The generic system simulated was similar to the model set-up for the Assasuni site in terms of hydrogeological conditions (e.g. 13m thick clay aquitard with K_x = 1x10^{-4} m/d, K_z = 1x10^{-5} m/d, 11m thick sandy aquifer with K_x = 0.2 m/d, K_z = 0.1 m/d, n_e = 0.25). The generic system is based on a design injection flow rate rather than a constant injection head. Use of a constant injection flow rate enables system parameters to be independently explored without the flow rate changing due to groundwater mounding or non-steady-state conditions. In the model an injection flow rate of 11.7 m^3/d was used to achieve a 52% design RE as recommended by Ward et al. (2008) (in this study, a 52% RE was shown to be quite reasonable for a recently established system and will only continue to increase with time) based on an extraction rate of 2 m^3/d. This extraction rate was calculated as the water demand required for a 100-person community (20 L per person/day, UN Water, N.D.). The generic model is not independent of site hydraulic conductivity and applies to systems where the injection and extraction wells are fully screened, and the aquifer is homogeneous and has an anisotropy ratio, K_x/K_z, of 2.

The injection wells-extraction well configuration was also modified from the Assasuni case, as this design has been shown to cause excessive extraction of ambient groundwater because of the poor location of the extraction well in an area under less influence from the injection wells. For the generic case, a single extraction well was located in the centre of the injection well area to reduce the chances of extracting ambient groundwater and was fully screened across the aquifer depth as this set-up was shown to enhance system performance (see Appendix F). Sensitivity simulations indicated that adopting a different well operating schedule (e.g., including a storage period or a shorter extraction period) had negligible impact on extraction TDS concentrations and overall system RE and therefore the same operating schedule of 120d injection period and 365d extraction period was adopted. Similarly, the impact of ambient groundwater TDS concentration had little effect on the system performance, provided TDS concentrations did not exceed 20 g/L and therefore an ambient groundwater TDS concentration of 4.95 g/L was applied in the model (see Appendix F for simulation results and a full discussion of generic model specifications).

To investigate the effect of the number and configuration of injection wells on the efficiency of an ASTR system in a non-dimensional framework, the parameter L_w^* was defined:

$$L_w^* = \frac{L_w}{\sqrt{\frac{Q_{net}T}{\pi B n_e}}} \qquad \text{Equation 3-1}$$

where T is the duration of the injection period [T] (120 days for the purposes of this modelling study), L_w is the lateral spacing between one injection well (shown in blue in Figure 26) and the extraction well (shown in red in Figure 26) [L] and Q_{net} [L^3/T] is the net injection rate per injection well calculated as: $Q_{net} = \frac{Q_{inj} - Q_{ext}}{number\ of\ injection\ wells}$. L_w^* represents the ratio of the spacing between one injection well and the extraction well to the theoretical radius of the injection plume (assuming a single injection well, see Figure 26).

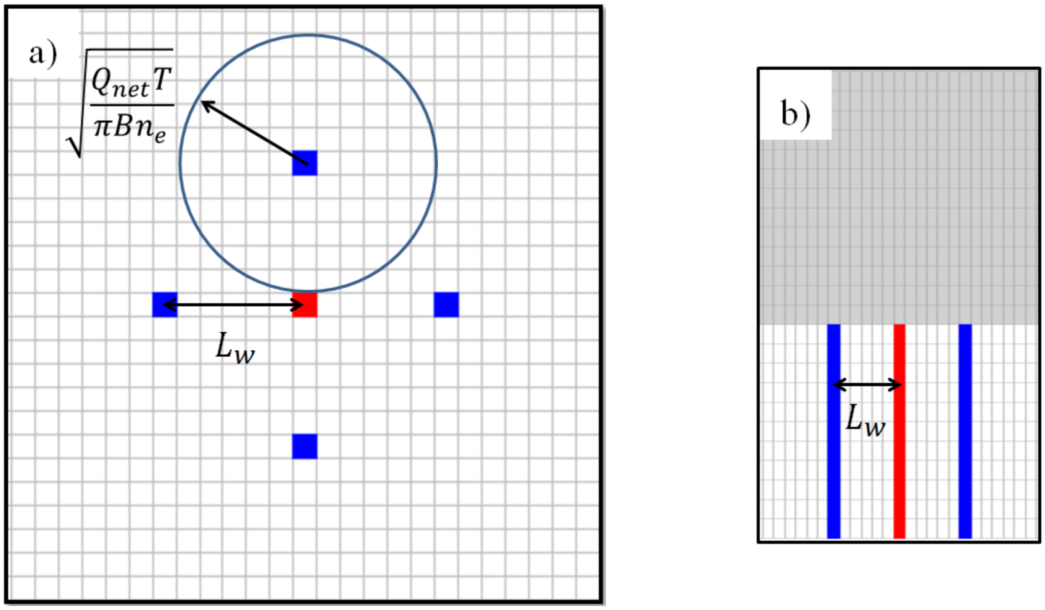

Figure 26: Generic model well spacing a) plan view and b) cross-section view (red cell indicates extraction well, blue cells indicate injection wells, and grey layers are aquitard layers).

Simulations were performed with the injection wells spaced at varying L_w^* (1, 0.5, 0.25) around the centre extraction well and in a variety of spatial configurations (3 wells in an equilateral triangle formation, 4 wells in a rectangular formation, 4 wells in a square formation and 6 wells in a rectangular formation). An ASTR system with four injection wells in a square formation around the extraction well at $L_w^* = 0.25$ was able to produce the design RE, as compared to systems with three or six wells (see Appendix F for detailed results). Additional simulations were performed with different L_w^* for ASTR systems of varying scale (Q_{net} = 2.4 m^3/d, Q_{net} = 12.1 m^3/d and Q_{net} = 24.3 m^3/d; four injection wells used). Regardless of the scale of the ASTR system, the highest RE (~48%) was obtained with L_w^* = 0.25 (Figure 27). (Note: system RE does not match design RE because the period of the start-up phase is not included in the calculation). The RE only marginally increased when using L_w^* = 0.25 compared to 0.5, however L_w^* = 0.25 is recommended to be used in ASTR system design as it requires smaller land space whilst still meeting the requirements for bacteria removal (retention time for the small-scale model with L_w^* = 0.25 is ~8 days, which exceeds the required aquifer retention time of 2.5 days). If a longer retention time were required, a spacing of L_w^* = 0.5 could be used and a similar RE would be achieved.

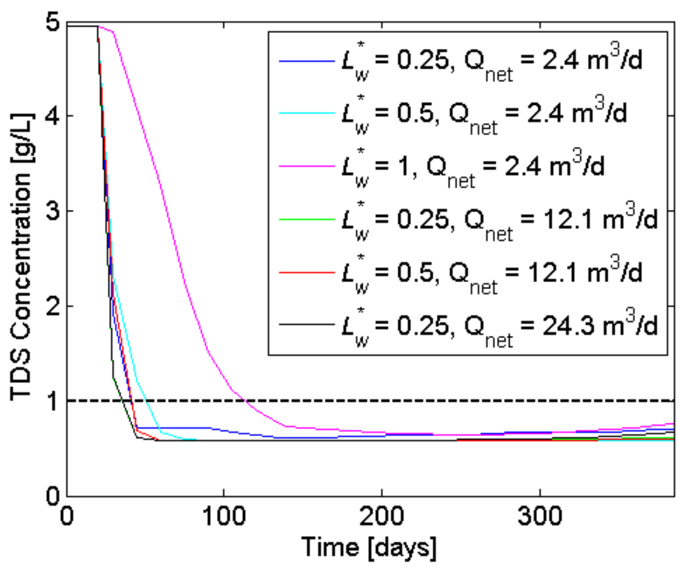

Figure 27: Effect of L_w^* on extraction TDS concentration (small-scale model (Q_{net} = 2.4 m^3/d), medium-scale model (Q_{net} = 12.1 m^3/d), and large-scale model (Q_{net} = 24.3 m^3/d)).

In an idealized system with a sharp brackish-freshwater interface L_w^* = 1 should provide a high RE, however, dispersive mixing significantly reduces the RE and therefore the recommended L_w^* (0.25) is less. Based on the work of Pavelic et al. (2002) and Ward et al. (2009), the RE will decrease as the width of the mixing zone increases relative to the theoretical radius of the injected freshwater. Following Pavelic et al (2002), a non-dimensional "relative dispersivity" (R^*_{Disp}) term can be defined:

$$R^*_{Disp} = \frac{\sqrt{\frac{Q_{net}T}{\pi B n_e}}}{\alpha_L} \qquad \text{Equation 3-2}$$

R^*_{Disp} represents the ratio of the theoretical plume radius to the longitudinal dispersivity. For the results presented in Figure 27 a constant R^*_{Disp}= 5.8 was used.

The critical R^*_{Disp} below which the RE is unacceptable was evaluated by testing different scale systems (small scale, Q_{net} = 2.4 m^3/d, medium scale, Q_{net} = 12.1 m^3/d, large scale,

Q_{net} = 24.3 m^3/d, and extra-large scale, Q_{net} = 36.4 m^3/d) with varying α_L. A similar relationship between R^*_{Disp} and RE values was exhibited for the different scale models: the RE approached zero when R^*_{Disp} was 1.5 or lower (Figure 28). For an ASTR system to be effective, R^*_{Disp} values should be greater than 1.5. Systems which do not meet this minimum criterion will not be able to produce water at the required 1g/L TDS concentration. The sharpness of the transition from 0% RE to a high RE decreases slightly as system scale increased and this should be explored in future studies.

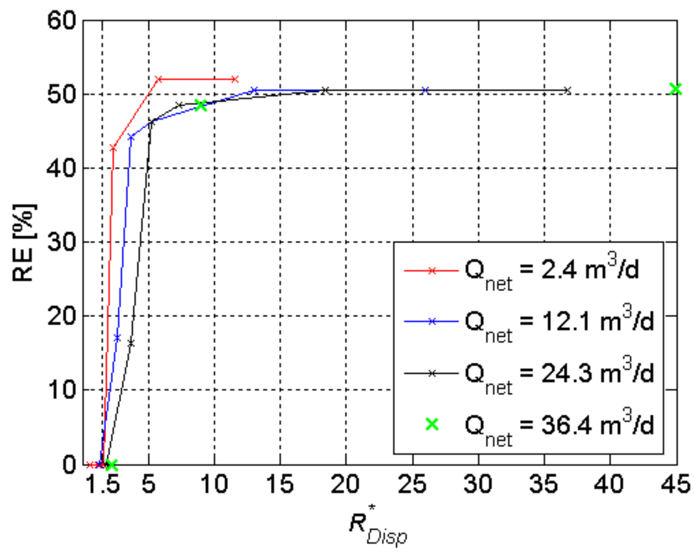

Figure 28: Effect of non-dimensionalized ratio R_{Disp}^* on RE for different scale models with 4 injection wells and L_w^* = 0.25.

The application of the non-dimensional parameters L_w^* and R^*_{Disp} in the design of ASTR systems was tested by simulating ASTR systems of different scales installed in different hydrogeological conditions (Table 4). Results indicate that varying parameters, such as Q_{net}, n_e, and B had minimal effect on the system RE and extraction water TDS concentrations provided L^*_w = 0.25 and R^*_{Disp} > 1.5 (Figure 29).

Table 4: Generic model parameters (pertinent to calculation of R_{Disp}^*), Generic base model parameters are in bold.

Generic Model	A	B	C	D	E	F
Effective Porosity, n_e [-]	**0.25**	0.4	0.25	0.1	0.1	0.1
Aquifer Thickness, B [m]	**11**	7	21	11	21	11

Generic Model Continued	A	B	C	D	E	F
Net Injection Rate, Q_{net} [m^3/d]	**2.4**	2.4	5	1.1	2	1.1
Longitudinal Dispersivity, α_L [m]	**2.5**	2.5	2.5	2.5	2.5	2.5
L_w^*	**0.25**	0.25	0.25	0.25	0.25	0.25
R_{Disp}^*	**2.4**	2.4	2.4	2.4	2.4	2.4

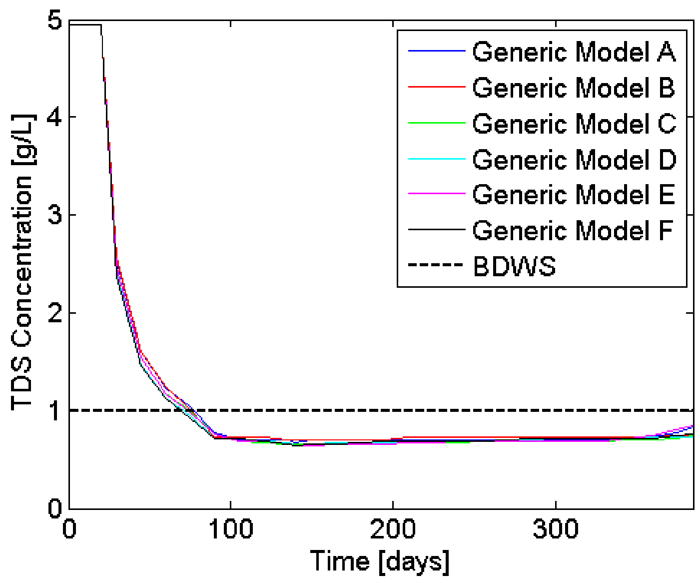

Figure 29: TDS concentration of extracted water for a 1 year period for models with different system parameters but all with 4 injection wells, L_w^* = 0.25 and R_{Disp}^* = 2.4.

Finally, the non-dimensional analysis presented thus far has not considered the influence of a regional hydraulic gradient on the system performance and design recommendations. A high regional hydraulic gradient will transport the injected water downstream of the injection area and this will lower the system RE. Simulations performed with varying hydraulic gradient at small and medium scales indicate that if the regional hydraulic gradient is less than i = 0.0025 the system will be able to meet and exceed the BDWS for the full extraction-only period (day 120 to day 365) without a modification to the system design (L_w^* = 0.25, R_{Disp}^* > 1.5, four injection wells, centre extraction well) (Figure 30). If the regional gradient were to exceed this limit, site design should be modified to include a downstream extraction well.

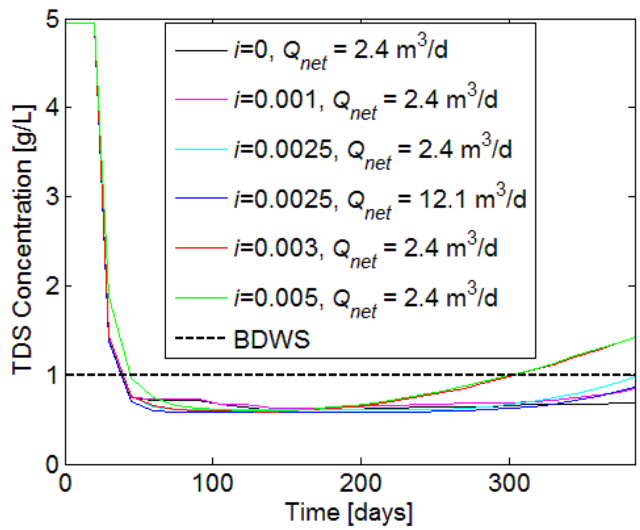

Figure 30: Effect of regional hydraulic gradient on extraction water TDS concentration.

3.4.4 Analysis of Assasuni Site Incorporating Non-dimensional Design Recommendations

The non-dimensinoal design recommendations, which showed that system efficiency was high when $L_w^* = 0.25$ and $R_{Disp}^* > 1.5$, were applied to the Assasuni site to determine the influence on system performance. Figure 31 shows the dramatic impact that incorporating these design recommendations has on extraction water TDS concentration. Without the design recommendations, the RE of the site is essentially 0%, however, once the site is designed with the non-dimensional parameters incorporated, the system RE is 53%.

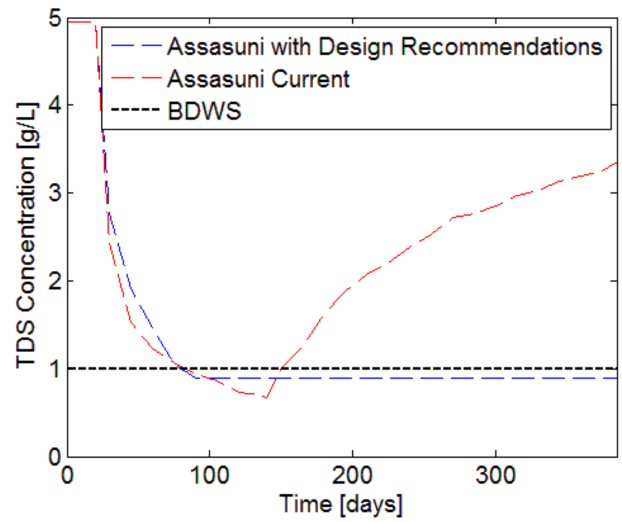

Figure 31: Influence of incorporating non-dimensional design parameters at the Assasuni site on extraction water TDS concentration.

From an engineering standpoint, the incorporation of these non-dimensional parameters makes the sites technically feasible and should therefore be incorporated into future site design.

3.5 Conclusions

This numerical study demonstrates that ASTR technology is a feasible alternative for providing drinking water to communities in south-western Bangladesh that currently face seasonal water scarcity issues provided the systems are design properly. A numerical model was first developed and calibrated based on the existing ASTR system at the Assasuni test site and simulation results show that a pocket of fresh groundwater will continue to expand in the aquifer around the injection area over a five year simulation period. Existing site design with an off-set extraction well is poor and causes undesirable extraction of ambient groundwater thereby reducing the system RE and promoting large fluctuations in the extraction water total dissolved solids (TDS) concentrations. As the systems currently implemented in Bangladesh are designed to achieve a constant injection head (rather than constant injection flow rate), sensitivity analyses showed that a system installed in an aquifer with higher transmissivity (higher K and/or B) will perform better as the injection rates and thus freshwater plume formed will be greater. If feasible, increasing the design injection head and/or number of injection wells will similarly improve the system performance. Other factors shown to influence the system performance, although to a lesser extent, include the aquifer dispersivity, spacing of injection wells and extraction rates.

At the Assasuni field site the regional hydraulic gradient is negligible but in settings with high hydraulic gradient (i > ~0.005), the RE will be reduced as the freshwater plume will migrate downstream of the extraction well and greater brackish-freshwater mixing will be induced. Tidal head fluctuations in the range of $\Delta H_{aquifer}$ = 0.3m are observed at the Assasuni site, however, simulations show that these fluctuations did not have a large impact on system efficiency although mixing did slightly increase due to enhanced groundwater flows. The ASTR systems are designed to be disaster resilient and thus to provide drinking water to communities following severe flooding events caused by, for example, cyclones and storm surges. The resiliency of the ASTR system was tested and from the simulations it may be expected that the systems will recover rapidly from a saltwater flooding inundation event during the injection phase (monsoon season), however, system RE was reduced to 0% until the subsequent injection phase if an inundation event occurred during the extraction-only phase (dry season).

From the simulation results, it can be concluded that future ASTR systems installed in Bangladesh should incorporate a design which maximizes injection head (e.g. hydraulically connecting water stored in a raised storage tank to the injection well) and includes an extraction well surrounded by injection wells spaced such that an aquifer retention time > 2.5 days is achieved for adequate bacteria removal. Further, if high extraction rates are required (e.g. 8 m^3/d), the system must be designed to also have a higher injection rate (i.e. ~12 m^3/d injected for 8 m^3/d extracted to yield the same net injection rate (0.96 m^3/d) as for the base case). If the ASTR system were to be installed in an area with high longitudinal dispersivity, low hydraulic conductivity, small aquifer thickness or a high regional hydraulic gradient, modifications to system design and operation should be made to maximize the injection rate (e.g. increasing injection head, number of wells and spacing).

To investigate the performance of an ASTR system more generally, a generic model was developed and the non-dimensional parameters ${L_w}^*$ and R^*_{Disp} were introduced so as to enable results to be applied to systems in different hydrogeological settings or with altered operational and design parameters. ${L_w}^*$, representing the ratio of the spacing between one injection well and the extraction well to the theoretical plume radius of one injection well, was tested at different scales (scale based on Q_{net}). An injection well spacing of ${L_w}^*$ = 0.25 with four injection wells surrounding a central extraction well design was shown to produce the best RE at all scales. The systems were then tested to determine the value of the longitudinal dispersivity, and thus the extent of fresh-brackish water mixing, that would lead to undesirable RE values. R^*_{Disp}, the ratio of the theoretical plume radius of one well to the longitudinal dispersivity in the system, was plotted against RE and the results showed that, at a variety of scales, the value of R^*_{Disp} should be greater than 1.5 to ensure that RE of the system was acceptable. Finally, for the conditions simulated it was shown that modifications to the system design are not required and these non-dimensional criteria can be applied if the regional hydraulic gradient is less than i = 0.0025 and the systems are installed in brackish aquifers with TDS concentration less than 20 g/L. Application of the non-dimensional design parameters applied at the Assasuni site resulted in a high (53%) RE. Future studies should investigate other factors impacting the system RE including anisotropy, heterogeneities, infiltration of saltwater from the surface, high-frequency time-varying injection rates and clogging.

3.6 References

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Chapter 4

4 CONCLUSIONS AND RECOMMENDATIONS

4.1 Summary and Conclusions

The need for disaster-resilient, water supply options is growing with more frequent natural disasters and higher population densities expected, both in Bangladesh and other regions experiencing water scarcity issues. Managed aquifer recharge (MAR) has the potential to be a leading alternative to other engineered systems such as dams and reservoirs for addressing global water crises by providing a reliable drinking water source at a low cost. Aquifer storage, transfer and recovery (ASTR) is a type of MAR where water is injected directly into the aquifer to create a pocket of unimpaired water that can be used as a water source. This thesis investigated the feasibility of community-scale ASTR for the south-western coastal region of Bangladesh, where water resources are vulnerable to the effects of natural disasters (e.g. cyclones, storm surges and seawater flooding). Further, this thesis explored the design and site selection for ASTR systems in a more generalized coastal setting. This study used a numerical model developed in SEAWAT-2005 to evaluate the impact of various hydrogeological and engineering design parameters on the performance of ASTR systems currently being installed in south-western Bangladesh. The model was first developed and then calibrated using data from the existing Assasuni ASTR field site in the Satkhira district of Bangladesh. The Assasuni site incorporates four gravity-fed injection wells with a single offset extraction well (pumped), where a mixture of pond and rainwater is injected into the aquifer for 120 days at an approximate TDS concentration of 0.6 g/L. The receiving aquifer is naturally brackish with TDS concentration ~5 g/L. Extraction takes place over a full 365 days to meet the needs of the surrounding community.

The system was evaluated on the basis of the ability to (i) yield extracted water at a total dissolved solids (TDS) concentration less than the Bangladesh Drinking Water Standard (BDWS) of 1 g/L; (ii) maximize recovery efficiency with the resources available (RE, the ratio of the volume of water extracted at TDS concentration < 1 g/L to the volume of water injected); and (iii) achieve an aquifer residence time of 2.5 days or more to ensure adequate removal of microbial contaminants from the extraction water (Rahman and Bhattacharya, 2006; Sidhu et al., 2010; Ward et al., 2009). As the ASTR system currently implemented in south-western Bangladesh is designed to achieve a constant injection head, sensitivity analyses showed that the aquifer hydraulic conductivity, aquifer thickness, injection head and injection well number and spacing significantly impacted the injection rate of freshwater into the aquifer and thus the system performance. Other hydrogeological parameters, including the aquifer dispersivity and presence of a regional hydraulic gradient were also shown to influence the system performance by altering the extent of brackish-freshwater mixing in the aquifer and downstream migration of the plume, respectively. Increasing the extraction rate from the system without an accompanying increase in injection rate caused large amounts of ambient groundwater to be withdrawn thereby decreasing system RE.

To guide ASTR system design in a diverse range of settings, a generic model was developed. The non-dimensional parameters L_w^* and R_{Disp}^* were introduced to be applied as criteria in the design of an ASTR system. Analysis performed at a variety of scales (Q_{net} = 2.4 m^3/d, 12.1 m^3/d or 24.3 m^3/d), showed that L_w^*, representing the ratio of the spacing between an injection well and the extraction well to the theoretical plume radius of one injection well, should be kept at 0.25 to maximize RE provided a minimum aquifer retention time of 2.5 days can be achieved. A spacing using L_w^* = 0.5 was also shown to have good RE and this design adjustment could be made to achieve a suitable aquifer retention time if required. The influence of R_{Disp}^* (the ratio of theoretical plume radius of one well to the aquifer longitudinal dispersivity) on RE was also examined and it was demonstrated that the system RE becomes unacceptable when R_{Disp}^* is less than 1.5. Systems should be designed such that the R_{Disp}^* value is kept above this value and for settings where this is not feasible (i.e. aquifer dispersivity it too great) then modifications to the ASTR system design must be considered.

4.2 Recommendations

This study has examined many features of the ASTR systems currently implemented in Bangladesh or the generic context; however, limitations of this study include:

- The numerical model was not used to examine the impacts of anisotropy or degrees of heterogeneity on the system performance. Anisotropy could lead to varying flow rates and patterns in the area surrounding the injection and extraction wells causing greater mixing between ambient groundwater and injectant or the migration of injected freshwater beyond the capture point of the extraction well, both resulting in reduced system RE. Heterogeneity could impact the system by causing larger movement of freshwater in high permeability layers and therefore greater mixing and migration, or the movement of water into lower hydraulic conductivity layers where water lost to residual volume surrounding porous media could cause reduced RE.
- The ASTR site in Assasuni is monitored by local personnel who are responsible for maintaining a constant water level in each injection well. The personnel in charge are not, however, aware of the implications of inconsistent injection heads and therefore injection wells frequently become dry at the surface and the infiltration rates are low for extended periods of time. The impacts of time-varying injection heads on the system efficiency should be examined in future studies and adjustments to site design or operation made accordingly.
- Clogging can be a major concern for ASTR sites (Bouwer, 2002; Pavelic et al., 2006). The impact on injection rates into the aquifer caused by progressive clogging should be monitored in future studies and guidelines on well rehabilitation developed to assist in site maintenance.
- Rainwater collected on rooftops or in ponds acts as the source water for these systems, however, in the event that a drought period occurs and existing pond water sources are required for injection, water with TDS concentrations exceeding the designed injection TDS concentration of ~0.6 g/L could be used as injectant. The impact on the system of using water with higher TDS concentration as injectant or simply leaving the system until such a time that water of the desired TDS concentration is available should be investigated.

- Current designs specify injection for 120 days either by general head boundary (base model) or injection well (generic model). This approach assumes that sufficient water is available to enable continuous injection over 120 days. Where this is not the case, modifications to system operation (e.g. modified injection schedules where injection occurs daily but for a limited number of hours) should be made to enhance system RE.
- The impacts on the system of an inundation event were explored and it was determined that an inundation event occurring during the dry season would result in 0% RE for the remainder of the dry period where only extraction is occurring. Methods of recovering the system, such as inducing high extraction rates on the system during the inundation period to rapidly remove injected saline water, should be investigated to determine feasibility and effectiveness in recovering system viability.
- Dispersivity is poorly quantified at the field sites in Bangladesh but has a large impact on the system RE. Tracer tests should be completed to determine aquifer dispersivity and modifications to the system (e.g. increasing injection rate into the aquifer) should be made to enhance RE if required.
- Models with a regional hydraulic gradient were investigated for both the Assasuni base model and the generic system. Additional investigations are required to determine if a downstream extraction well would improve RE and if so what the location of the extraction well should be.
- A full cost-benefit and social analysis of the ASTR technology implementation should be performed to ensure that these systems meet the needs of the surrounding community in the most effective way possible
- Some previous ASTR studies have shown that the injection of freshwater into the aquifer can cause dissolution of minerals and undesirable water quality changes (e.g., increase in heavy metal concentrations) (Pavelic et al., 2006). The water quality of the extracted water is routinely monitored for the ASTR system in Bangladesh, however, the potential for long-term geochemical changes in the aquifer should be evaluated.

- Current system design uses gravel-packed injection wells to pass injection water from the surface to the aquifer. The use of local materials may result in materials other than gravel (though still high permeability) to be used for the injection well creating variation in the injection well conductance. The influence of the conductance of the material on the system performance should be established.

While the generic model was developed to guide site selection and design criteria in a variety of settings, it is recommended that future sites installed in Bangladesh adopt a spacing parameter of $L_w{}^*$ = 0.25 and maintain R^*_{Disp} > 1.5. Site design and selection are critical for ensuring the successful performance of a system and also so that costs can be minimized in order to continue developing new sites and maintaining existing ones. The recommendations provided in this thesis should be used to guide future and existing ASTR projects.

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APPENDICES

Appendix A – Assasuni ASTR Site

The Assasuni ASTR site is located in the Satkhira district of south-western coastal Bangladesh (Figure A1) and began operation in 2011. The site is within 200m of a tidally influenced river (shown in satellite photo, Figure A1) and experiences diurnal hydraulic head fluctuations in the aquifer (~0.3 m amplitude). The ASTR installation is accessed by approximately 175 registered citizens but, as it is centrally located in Assasuni, it is estimated that upwards of 1000 people access the site on a daily basis (Ahmed, 2012).

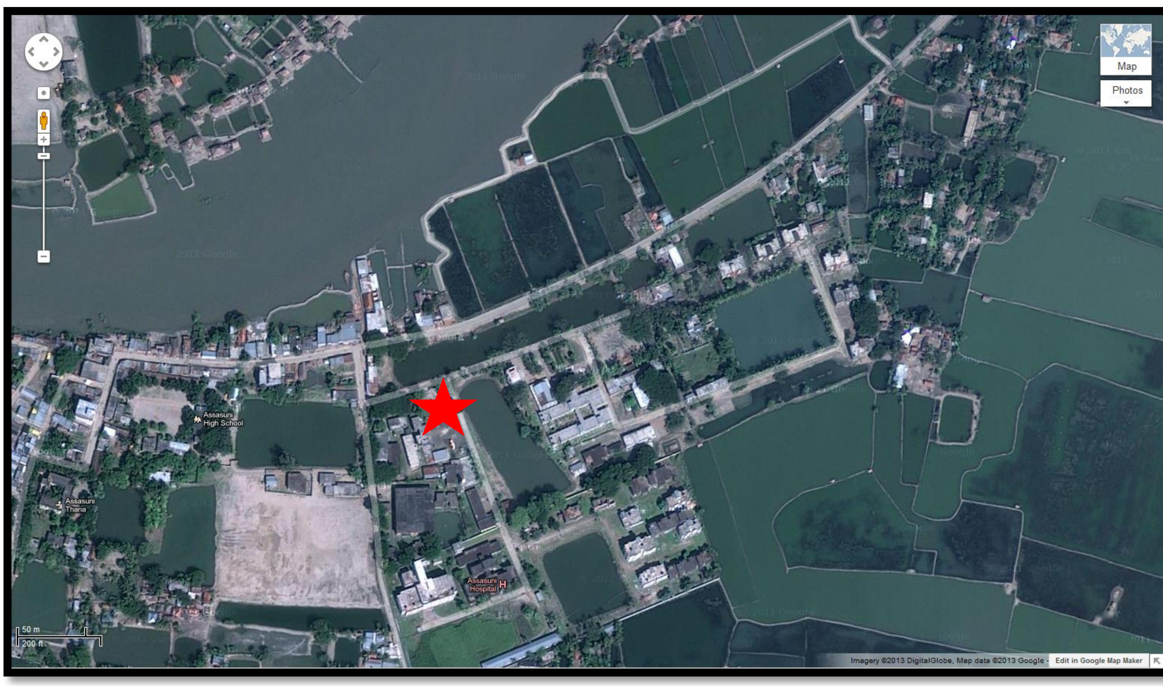

Figure A1: Assasuni ASTR site satellite photograph (GoogleMaps, 2013).

The site incorporates four gravity-fed injection wells which inject a combination of rain and pond water, one extraction well offset from the injection well area by approximately 0.5m, and a water holding tank with a slow sand filter (Figure A2). A thick clay layer (13m) overlies a sandy aquifer (11m) at the Assasuni site (Figure A3). The ambient total dissolved solids (TDS) concentration in the aquifer is ~5 g/L. The objective of the ASTR system is to reduce the TDS concentration of the extracted groundwater to the Bangladesh Drinking Water Standard (BDWS) 1 g/L (Rahman and Bhattacharya, 2006).

The low permeability clay layer is bypassed by the injection wells and the freshwater is injected directly into the aquifer via a screened, high permeability gravel zone.

Figure A2: Assasuni field site well layout (square concrete, raised structures are injection wells, metal hand pump in upper-left corner is the extraction well).

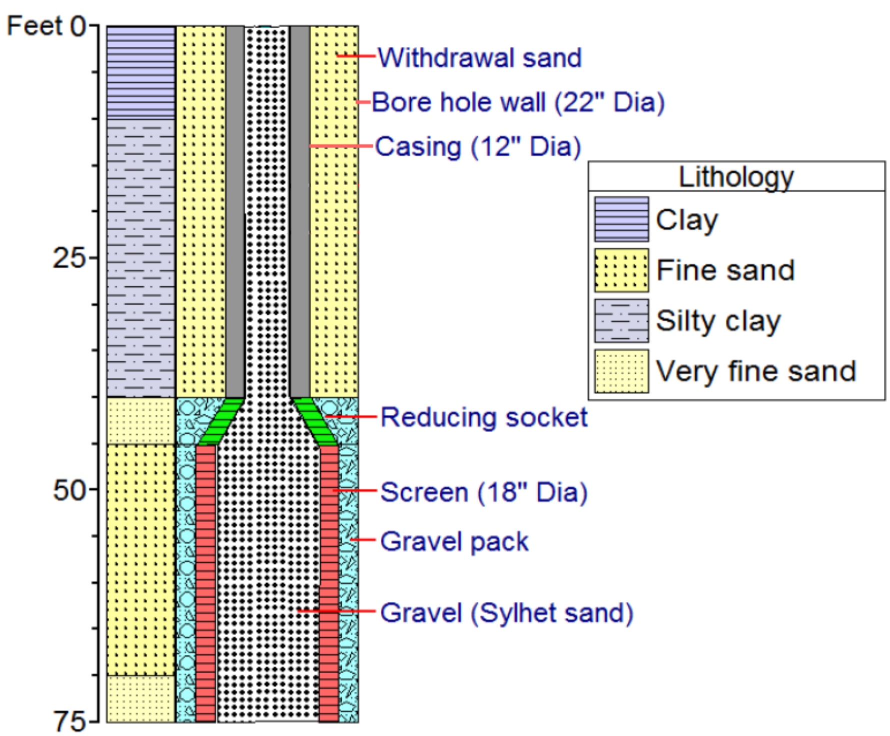

Figure A3: Assasuni site borehole log showing individual injection well configuration (Hasan, 2012).

The description of the numerical model representing the Assasuni site was presented in Chapter 3. To best represent the field set-up, the base model presented has 4 injection wells offset from an extraction well (Figure A4). Two injection wells have a diameter of 0.3048m and the remaining two injection wells have a diameter of 0.5588m. One well of each diameter is screened from *z* = -13m to *z* = -14m, while the other is screened from *z* = -13m to *z* = -24m.

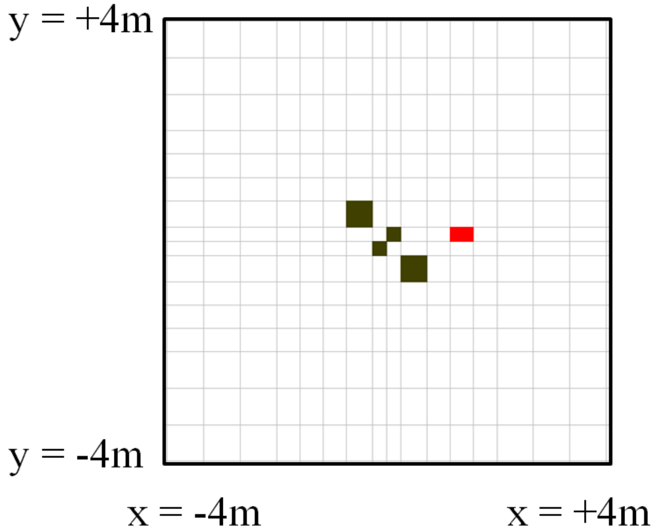

Figure A4: Assasuni model well layout (green cells are injection wells, red cell is extraction well).

Appendix B – Governing Equations for SEAWAT-2005

This numerical study uses SEAWAT-2005, a variable density groundwater flow and solute transport code. In SEAWAT-2005 transient movement of freshwater through an anisotropic, heterogeneous saturated aquifer is described using the three-dimensional groundwater flow equation (Equation B-1) and the advection-dispersion-reaction equation (Equation B-2) (Guo and Langevin, 2002).

$$S_s \frac{\delta h}{\delta t} = \frac{\delta}{\delta x}\left(K_x \frac{\delta h}{\delta x}\right) + \frac{\delta}{\delta z}\left(K_y \frac{\delta h}{\delta y}\right) + \frac{\delta}{\delta z}\left(K_z \frac{\delta h}{\delta z}\right) + R^* \qquad \text{Equation B-1}$$

where S_s is the specific storage [1/L], $\frac{\delta h}{\delta t}$ is the change in hydraulic head, h, [L] with time, t, [T], K_i is the hydraulic conductivity in the i-plane where i is the x, y or z-direction [L/T], $\frac{\delta h}{\delta i}$ is the change in hydraulic head with space in the i-plane where i is the x, y or z-direction [-], and R^* is the source or sink term [1/T].

$$\frac{\delta C}{\delta t} = D_x \frac{\delta^2 C}{\delta x^2} + D_y \frac{\delta^2 C}{\delta y^2} + D_z \frac{\delta^2 C}{\delta z^2} - v_x \frac{\delta C}{\delta x} - v_y \frac{\delta C}{\delta y} - v_z \frac{\delta C}{\delta z} \qquad \text{Equation B-2}$$

where $\frac{\delta C}{\delta t}$ is the change in concentration with time [M/L^3/T], D_i is the coefficient of hydrodynamic dispersion in the i-plane where i is the x, y or z-direction [L^2/T], $\frac{\delta^2 C}{\delta i^2}$ is the change in concentration in the i-plane where i is the x, y or z-direction [M/L^3/L^2], and v_i is the fluid velocity in the i-plane where i is the x, y or z-direction [L/T].

With the interaction of freshwater and brackish water, density differences must be taken into account. This is done by first incorporating the concept of equivalent freshwater head, where head values in a saline environment are converted using Equation B-3 to head values in a corresponding freshwater environment (Guo and Langevin, 2002). The equivalent fresh water head, h_f, [L] is given by:

$$h_f = \frac{\rho h}{\rho_f} - \frac{\rho - \rho_f}{\rho_f} Z \qquad \text{Equation B-3}$$

where h is the head [L], ρ is the density of the water in the saline aquifer [M/L^3], ρ_f is the density of freshwater [M/L^3], and Z is the elevation.

Darcy's Law (Equations B-4 to B-6) describes the movement of a fluid through a porous media due to a pressure gradient (Guo and Langevin, 2002).

$$q_x = -\frac{k_x}{\mu}\frac{\delta P}{\delta x}$$ Equation B-4

$$q_y = -\frac{k_y}{\mu}\frac{\delta P}{\delta y}$$ Equation B-5

and

$$q_z = -\frac{k_z}{\mu}\left(\frac{\delta P}{\delta z} + \rho g\right)$$ Equation B-6

where q_i is the specific discharge in the i-plane where i is the x, y or z-direction [L/T], k_i is the intrinsic permeability in the i-plane where i is the x, y or z-direction [L^2], μ is the fluid viscosity, $\frac{\delta P}{\delta i}$ is the change in pressure with space in the i-plane where i is the x, y or z-direction, and g is the acceleration due to gravity [L/T^2].

It can be assumed that the principal axes of permeability align with the coordinate system used by these equations. Once Darcy's law is simplified by this assumption and then written in terms of equivalent freshwater head, the equations can be substituted into the groundwater flow equation to account for density differences (Guo and Langevin, 2002).

One further adjustment must be made to the groundwater flow equation before it can effectively predict the movement of fluids in a variable-density environment. The relationship between saltwater density and solute concentration must be considered (Guo and Langevin, 2002). Baxter and Wallace (1916) developed the empirical formula describing this relation (Equation B-7):

$$\rho = \rho_f + \frac{\delta \rho}{\delta C}C$$ Equation B-7

where $\frac{\delta\rho}{\delta C}$ is the change in density with concentration and is set to the standard 0.7143 (Baxter and Wallace, 1916; Guo and Langevin, 2002).

These substitutions yield the variable density groundwater flow equation (Guo and Langevin, 2002):

$$\frac{\delta}{\delta x}\left(\rho K_{fx}\left[\frac{\delta h_f}{\delta x}+\frac{\rho-\rho_f}{\rho_f}\frac{\delta Z}{\delta_x}\right]\right)+\frac{\delta}{\delta y}\left(\rho K_{fy}\left[\frac{\delta h_f}{\delta y}+\frac{\rho-\rho_f}{\rho_f}\frac{\delta Z}{\delta_y}\right]\right)+\frac{\delta}{\delta z}\left(\rho K_{fz}\left[\frac{\delta h_f}{\delta z}+\frac{\rho-\rho_f}{\rho_f}\frac{\delta Z}{z}\right]\right)=\rho S_f\frac{\delta h_f}{\delta t}+0.7143n_e\frac{\delta C}{\delta_t}+R^*$$ Equation B-8

Appendix C – Model Domain Size and Grid Independence Tests

Rigorous numerical testing was performed to ensure that the numerical model solution was converged. The numerical solver scheme for the advection-dispersion equation, grid discretization and the size of the model domain was tested. Two different solver schemes for the advection-dispersion equation were investigated: (i) hybrid method of characteristics (HMOC) which couples the method of characteristics (MOC) and modified method of characteristics (MMOC) schemes depending on the sharpness of the advection front, and (ii) total variation diminishing (TVD) scheme.

Different sizes of the model domain were tested to determine the distance the model boundaries had to be located from the simulated ASTR system so as to not affect the numerical results (Figure C1a). The conditions (constant head) imposed at the external model boundaries affect the simulation results if the boundaries are too close to the injection area as they will induce artificial flows between the injection wells and the boundary. Model domain sizes of 200m through 1000m (increasing size by 200m per model) were examined.

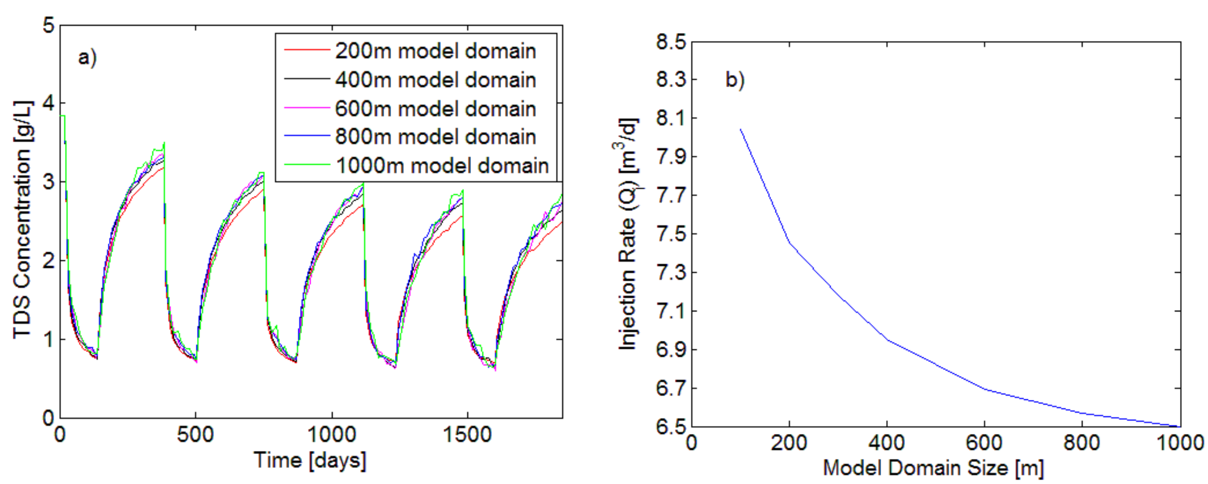

Figure C1: a) Extraction concentration for different model domain sizes using the HMOC solver (concentration observations taken at the extraction well, x = +1.5m) and b) impact of model domain size on injection rate (HMOC scheme used).

While concentrations results were affected by the size of the model, the injection rates were more significantly altered (Figure C1b). The 1000m model had the least impact on injection rates, however, as model domain size increases so too does the computational effort. The 800m x 800m model domain size was selected as this struck a balance between required computational effort and the effect of the external boundary conditions on the injection rate.

Varying sizes of the model domain were also tested using the TVD solution scheme (Figure C2a). The simulated concentrations exhibited small fluctuations when the TVD scheme was used. A closer comparison between the final 800m model for both HMOC and TVD schemes (Figure C2b) shows higher numerical instability for the TVD model and therefore the HMOC scheme was adopted for all models.

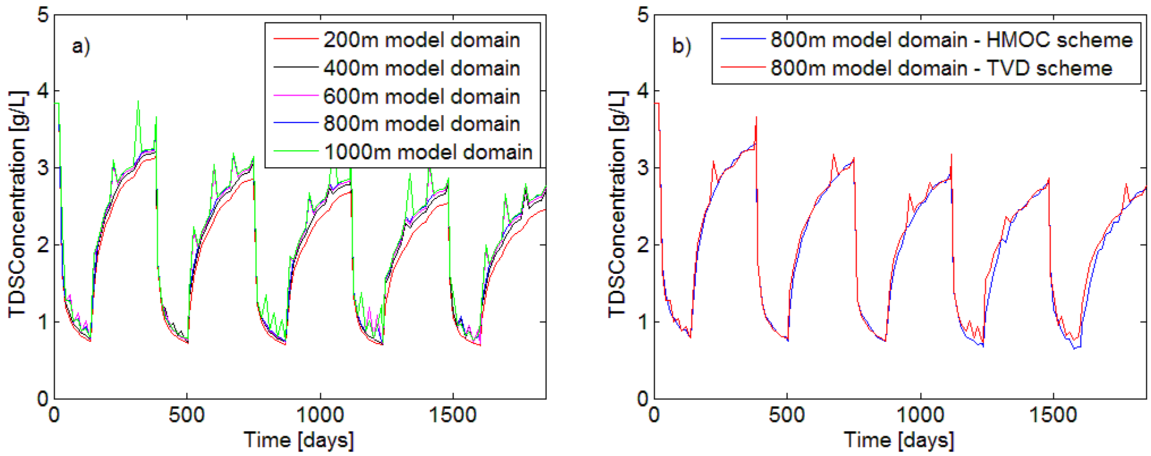

Figure C2: a) Extraction concentration for different model domain sizes using the TVD solver (concentration observations taken at the extraction well, x = +1.5m), and b) extraction concentration for solvers HMOC versus TVD scheme for 800x800 model domain (concentration observations taken at the extraction well, x = +1.5m).

Grid discretization tests were also performed to ensure the model solution was converged. Simulations using uneven grid spacing, 50m even grid spacing and 25m grid spacing were conducted to determine the sensitivity of the results to the grid size (Figure C3).

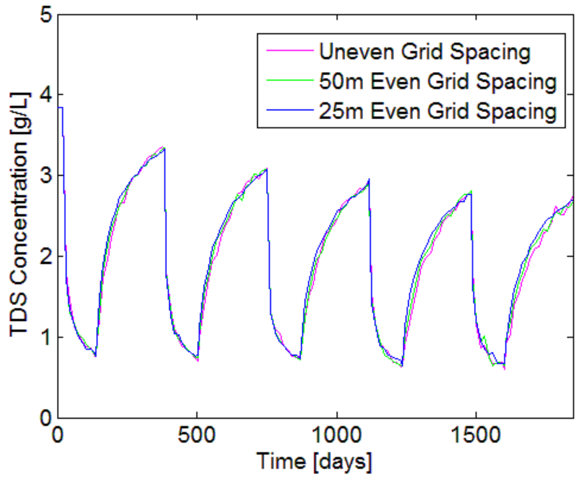

Figure C3: Grid discretization test results.

The difference between the model results for these grid spacing was negligible, however, mass balance error in the model decreased as grid discretization increased. The mass balance error must be minimized to ensure that mass is not artificially leaving or entering the model without a source or sink. The uneven grid spacing previously discussed had an average mass balance error of 15% per time step, while the 50m even grid spacing had an average error of 10% per time step. The average mass balance error for the 25m even grid spacing was approximately 4% per time step. A 25m grid spacing was therefore chosen so as to reduce mass balance error while still maintaining a reasonable computational time. A mass balance error of around 3.3% per time step for the HMOC scheme is considered reasonable (Konikow, 2011).

Appendix D – Spatial Moments Analysis

Spatial moments analysis was used to quantify the time-varying characteristics of the freshwater plume in the system. Spatial moments typically quantify the distribution of solute in the system, however, for our case where freshwater is injected into a brackish aquifer, the spatial moments equations are modified to characterize the zone of lower solute concentration. Moments were calculated for a two-dimensional slice of the model domain as variation in the radial direction was negligible. Calculation of two- rather than three-dimensional moments also reduced the computation demand of the calculations.

The zeroth spatial moment (M_{00}) represents the mass of freshwater in a two-dimensional aquifer slice at a given time (t) and is calculated via:

$$M_{00} = \int\int n_e[C_{amb} - C(x,z,t)]dxdz \qquad \text{Equation D-1}$$

where n_e is the effective porosity [L^3/L^3], C_{amb} is the ambient groundwater concentration [M/L^3], $C(x,z,t)$ is the concentration in a particular cell [M/L^3] at a given t, dx is the width of the particular cell [L] and dz is the height of the particular cell [L].

The first spatial moment, once normalized, gives the centroid of the fresh water plume at a given time:

$$M_{10} = \int\int n_e[C_{amb} - C(x,z,t)]xdxdz \qquad \text{Equation D-2}$$

$$M_{01} = \int\int n_e[C_{amb} - C(x,z,t)]zdxdz \qquad \text{Equation D-3}$$

$$x_c = \frac{M_{10}}{M_{00}}, z_c = \frac{M_{01}}{M_{00}} \qquad \text{Equation D-4}$$

where M_{10} is the first spatial moment in the x-direction, M_{01} is the first spatial moment in the z-direction, x is the x-coordinate of the particular cell [L], z is the z-coordinate of the particular cell [L], x_c is the x-coordinate of the fresh water plume centroid [L], and z_c is the z-coordinate of the fresh water plume centroid [L].

The second spatial moment, once normalized, gives the variance of the freshwater plume at a given time:

$$\bar{M}_{20} = \int \int n_e[C_{amb} - C(x,z,t)](x - x_c)^2 dxdz \qquad \text{Equation D-5}$$

$$\bar{M}_{02} = \int \int n_e[C_{amb} - C(x,z,t)](z - z_c)^2 dxdz \qquad \text{Equation D-6}$$

$$\sigma_{xx}^2 = \frac{\bar{M}_{20}}{M_{00}}, \sigma_{zz}^2 = \frac{\bar{M}_{02}}{M_{00}} \qquad \text{Equation D-7}$$

where $\bar{M}_{20}$ is the second spatial moment in the *x*-direction [], $\bar{M}_{02}$ is the second spatial moment in the *z*-direction [], σ_{xx}^2 is the variance in the *x*-direction [-], and σ_{zz}^2 is the variance in the *z*-direction [-].

Appendix E – Additional Sensitivity Analyses Results

This appendix provides additional sensitivity analyses results that were performed on the Base model representing the Assasuni site. These models are included in an appendix as they did not have a large impact on TDS concentrations and overall system efficiency relative to the analyses completed in Chapter 3. Table E1 provides information on all the sensitivity analyses which were conducted. For each sensitivity test performed, the injection rate, the freshwater plume radius, the mass of freshwater in a two-dimensional slice, the (x,z) centroid of the plume and the plume's variance are recorded. Larger injection rates typically led to larger freshwater plume radii and mass of freshwater in the system and the BDWS was achieved more rapidly and maintained than models with lower injection rates. The position of the plume centroid indicates the shift in the plume in the x and z-directions. If the plume centroid remains close to x = 0m, z = -19m, there is limited plume drift and RE values should remain high excluding other factors. Variance of the plume, represented by σ_{xx}^2 and σ_{zz}^2, gives an indication of the extent of mixing between brackish groundwater and freshwater occurring in the system. Higher values of either σ_{xx}^2 or σ_{zz}^2 indicate a greater amount of brackish-freshwater mixing is occurring in the x or z-directions than for other models.

Table E1: Comprehensive sensitivity analyses results.

Model	Sensitivity Value	Injection Rate [m^3/d]		Plume Radius [m]	Fresh Water Mass [kg]	Plume Centroid (*x*,*z*)	Mixing σ_{xx}^2	σ_{yy}^2
Aquifer Hydraulic Conductivity (*K*)	K_x = 0.1 m/d , K_z = 0.05 m/d	2.88	**120 d**	0.8	78	(-0.2, -18.5)	22.5	10.4
			1 yr	0.0	49	(-0.2, -19.5)	26.9	9.2
			5 yr	0.0	143	(-0.5, -19.3)	65.0	9.7
	K_x = 0.2 m/d, K_z = 0.1 m/d	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	K_x = 1.0 m/d, K_z = 0.5 m/d	20.57	**120 d**	9.9	440	(-0.1, -18.4)	144.7	10.6
			1 yr	8.4	399	(-0.3, -18.3)	134.7	10.6
			5 yr	24.0	1900	(-0.4, -18.1)	675.8	10.6
	K_x = 5.0 m/d, K_z = 2.5 m/d	99.12	**120 d**	25.5	2080	(0.1, -18.3)	738.7	10.7
			1 yr	27.1	2030	(0, -18)	729.0	10.6
			5 yr	56.3	15800	(-0.2, -18)	4419.9	11.1
	K_x = 10.0 m/d, K_z = 5.0 m/d	197.26	**120 d**	35.4	4650	(0.3, -18.3)	1623.1	10.8
			1 yr	35.4	4670	(0, -17.9)	1663.2	10.8
			5 yr	387.5	36600	(0, -18)	7728.2	11.5
Aquifer Longitudinal Dispersivity (α_L)	1.0	4.85	**120 d**	5.5	125	(-0.3, -18.7)	25.8	9.8
			1 yr	2.8	86	(-0.3, -19.3)	22.3	8.9
			5 yr	9.4	264	(-0.9, -19)	76.7	9.4
	2.5	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	5.0	4.86	**120 d**	0.0	119	(-0.2, -18.3)	50.0	10.9
			1 yr	0.0	88	(-0.4, -18.8)	60.2	10.9

Table E1 Continued: Comprehensive sensitivity analyses results.

			5 yr	0.0	311	(-0.8, -18.7)	165.4	11.1
	10.0	4.87	**120 d**	0.0	116	(-0.2, -18.2)	71.6	11.2
			1 yr	0.0	90	(-0.4, -18.6)	97.0	11.5
			5 yr	0.0	353	(-0.7, -18.4)	291.0	11.9
Aquifer Thickness (*B*)	5 m	2.71	**120 d**	4.3	56	(-0.6, -15.4)	30.8	2.4
			1 yr	0.0	22	(-0.8, -15.3)	42.5	3.0
			5 yr	0.0	58	(-0.9, -15.2)	72.8	3.0
	11 m	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	20 m	6.26	**120 d**	8.6	160	(-0.2, -19.7)	31.5	17.2
			1 yr	1.2	121	(-0.4, -20.4)	31.5	16.5
			5 yr	18.8	420	(-0.9, -21)	86.8	21.9
	30 m	6.79	**120 d**	10.2	177	(-0.2, -19.9)	33.0	17.8
			1 yr	4.2	135	(-0.4, -20.5)	32.2	16.8
			5 yr	20.9	474	(-1, -21.2)	88.5	23.3
	40 m	7.08	**120 d**	11.0	183	(-0.2, -20)	34.2	18.0
			1 yr	5.5	140	(-0.4, -20.5)	32.7	16.8
			5 yr	23.0	535	(-1, -23.4)	86.2	78.6
Hydraulic Gradient (*i*), flowing from y = -400 m boundary to y = +400 m boundary	**0**	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	0.0005	4.74	**120 d**	3.5	122	(-0.2, -18.5)	31.0	10.3
			1 yr	0.0	85	(-0.2, -19.2)	34.4	9.9
			5 yr	5.1	282	(-0.2, -18.9)	107.1	10.3
	0.0015	4.75	**120 d**	1.1	120	(0, -18.5)	37.0	10.6
			1 yr	0.0	86	(0.1, -19.1)	42.2	10.3
			5 yr	1.1	302	(1.4, -18.8)	141.6	10.8
	0.0025	4.75	**120 d**	1.1	120	(0.1, -18.5)	36.9	10.6

Table E1 Continued: Comprehensive sensitivity analyses results.

			1 yr	0.0	86	(0.4, -19.1)	42.3	10.2
			5 yr	1.1	307	(2.9, -18.8)	141.1	10.8
	0.005	4.74	**120 d**	1.4	120	(0.4, -18.5)	37.1	10.6
			1 yr	0.0	88	(1.3, -19.1)	43.1	10.1
			5 yr	1.5	348	(7.3, -18.8)	150.6	10.6
Injection Head (*H*)	**1.58 m**	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	2.58 m	6.92	**120 d**	5.1	167	(-0.2, -18.5)	45.2	10.4
			1 yr	1.6	127	(-0.4, -18.9)	44.0	10.3
			5 yr	12.0	468	(-1, -18.7)	143.4	10.0
	3.58 m	8.98	**120 d**	5.9	211	(-0.2, -18.4)	61.7	10.5
			1 yr	3.1	169	(-0.4, -18.7)	57.5	10.5
			5 yr	15.6	662	(-0.8, -18.5)	209.0	10.3
	4.58 m	11.05	**120 d**	5.9	254	(-0.1, -18.4)	80.3	10.6
			1 yr	3.9	212	(-0.4, -18.6)	74.3	10.6
			5 yr	17.7	868	(-0.8, -18.4)	292.8	10.5
Extraction Rate (Q_{ext})	0.5 m^3/d	4.43	**120 d**	1.4	122	(-0.1, -18.4)	37.8	10.6
			1 yr	0.0	102	(-0.3, -18.7)	37.8	10.5
			5 yr	9.9	367	(-0.7, -18.6)	115.5	10.2
	1.029 m^3/d	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	2.0 m^3/d	5.63	**120 d**	3.5	121	(-0.4, -18.6)	31.2	10.2
			1 yr	0.0	68	(-0.4, -19.6)	35.0	9.2
			5 yr	0.0	196	(-1, -19.3)	96.1	10.0
	3.0 m^3/d	6.44	**120 d**	3.1	121	(-0.4, -18.7)	32.1	10.2
			1 yr	0.0	57	(-0.3, -19.8)	36.8	8.8
			5 yr	0.0	143	(-0.8, -19.5)	90.0	9.8

Table E1 Continued: Comprehensive sensitivity analyses results.

	5.0 m^3/d	8.06	**120 d**	3.1	122	(-0.4, -18.7)	33.4	10.2
			1 yr	0.0	42	(-0.1, -20.2)	38.2	8.1
			5 yr	0.0	84	(-0.7, -19.8)	76.7	9.4
	8.0 m^3/d	10.48	**120 d**	3.1	125	(-0.3, -18.8)	34.2	10.2
			1 yr	0.0	28	(0.2, -20.4)	39.0	7.3
			5 yr	0.0	44	(-0.3, -20.1)	58.7	9.0
Number and Spacing of Injection Wells (well diameter = 0.3048m)	1 well	3.73	**120 d**	5.8292	91	(0.1, -18.8)	27.6	9.9
			1 yr	0	61	(0.1, -19.5)	33.3	9.2
			5 yr	4.6886	193	(-0.3, -19.2)	86.4	9.8
	2 wells	4.27	**120 d**	2.9	103	(-0.4, -18.7)	30.7	10.0
			1 yr	0.0	72	(-0.4, -19.3)	35.6	9.7
			5 yr	4.1	237	(-1, -19)	99.8	10.1
	4 wells, 0.6m spacing (square)	4.68	**120 d**	3.3	111	(-0.4, -18.6)	33.0	10.1
			1 yr	0.0	78	(-0.5, -19.2)	37.3	9.9
			5 yr	5.3	266	(-1.1, -18.9)	106.6	10.2
	4 wells, 1.8m spacing (square)	4.87	**120 d**	3.7	117	(-0.3, -18.6)	35.5	10.1
			1 yr	0.0	85	(-0.4, -19.1)	38.5	10.1
			5 yr	5.6	312	(-1.2, -18.9)	114.6	10.1
	4 wells, 0.6m spacing (line)	4.87	**120 d**	7.3917	110	(0.1, -18.6)	31.6	10.0
			1 yr	0	77	(0, -19.2)	35.7	9.9
			5 yr	11.298	271	(-0.5, -19)	105.3	10.1
	6 wells, 0.6 m spacing (rectangle)	4.94	**120 d**	7.3917	114	(-0.4, -18.6)	32.6	10.0
			1 yr	0	81	(-0.5, -19.2)	36.5	10.0
			5 yr	12.2095	284	(-1.1, -18.9)	109.6	10.2
Depth of Extraction Well Screen	**z = -13m to -16m**	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	z = -15m to -18m	4.85	**120 d**	3.1	124	(-0.3, -18.5)	30.7	10.6
			1 yr	0.0	81	(-0.6, -19)	32.4	11.2

Table E1 Continued: Comprehensive sensitivity analyses results.

			5 yr	4.8	260	(-1.5, -18.8)	85.7	10.7
	z = -18m to -21m	4.84	**120 d**	3.5	124	(-0.3, -18.3)	30.9	10.8
			1 yr	0.0	80	(-0.6, -17.9)	33.0	11.6
			5 yr	6.5	257	(-1.6, -17.8)	84.3	11.0
	z = -21m to -24m	4.84	**120 d**	3.5	123	(-0.3, -18.2)	31.1	10.1
			1 yr	1.6	87	(-0.3, -17.4)	33.0	9.0
			5 yr	7.8	285	(-0.8, -17.5)	103.3	9.5
	z = -13m to -24m	4.83	**120 d**	1.9	122	(-0.2, -18.3)	37.1	10.7
			1 yr	0.0	78	(-0.7, -18.2)	44.2	11.2
			5 yr	1.8	274	(-1.6, -18.2)	120.0	11.1
Season of Extraction (Days of Injection/Days of Extraction)	120d/245d	4.02	**120 d**	1.1	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.3	283	(-0.9, -18.9)	99.6	10.1
	120d/365d	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
Well Diameter	0.1m	4.48	**120 d**	4.9	111	(-0.5, -18.8)	24.0	9.5
			1 yr	1.7	75	(-0.4, -19.5)	23.1	8.8
			5 yr	7.7	237	(-0.9, -19.1)	73.5	9.4
	0.3048m	4.92	**120 d**	4.3	119	(-0.3, -18.6)	30.0	10.0
			1 yr	0.0	84	(-0.3, -19.2)	33.1	9.9
			5 yr	6.4	283	(-0.9, -18.9)	102.8	10.1
	0.5588m	5.18	**120 d**	3.4	123	(0, -18.5)	34.3	10.3
			1 yr	0.0	89	(-0.1, -19.1)	35.5	10.0
			5 yr	7.6	304	(-0.8,-18.9)	107.8	10.1
Location of Extraction Well	**x = +0.5m**	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	x = +3.065m	4.71	**120 d**	3.7	125	(-0.2, -18.4)	36.7	10.7

Table E1 Continued: Comprehensive sensitivity analyses results.

			1 yr	0.0	97	(-0.4, -18.8)	39.2	10.8
			5 yr	8.6	308	(-1.6, -18.7)	125.3	10.8
Tidal Effects	$\Delta H_{aquifer}$ = 0m	4.85	**120 d**	3.6	123	(-0.3, -18.5)	31.3	10.3
			1 yr	0.0	87	(-0.4, -19.2)	33.3	9.9
			5 yr	6.4	283	(-0.9, -18.9)	99.6	10.1
	$\Delta H_{aquifer}$ = 0.3m	7.47	**120 d**	3.68	133	(-0.2, -18.4)	39.0	10.6
			1 yr	0	142	(-0.4, -18.7)	60.4	10.7
			5 yr	-	-	-	-	-

Analysis of the influence of the number and configuration of injection wells on the ASTR system performance was presented in Chapter 3. Supplementary models were run with 1 injection well, 4 injection wells in a line configuration and 6 injection wells in a rectangular configuration. Limited increases in the injection rate occurred when 6 injection wells were implemented rather than 4 and thus the mass of freshwater in the aquifer did not increase greatly. Implementing 4 wells rather than 1 or 2, did improve extraction TDS concentrations significantly and 4 wells are recommended for future site design, however, 4 wells in a line configuration did not improve the system to a great extent, so a square configuration is recommended (Figure E1). The shift in the freshwater plume and the extent of mixing in the *x* and *z*-directions was consistent between models with different number of injection wells. Future systems to be installed in a similar hydrogeological setting can be effective with four injection wells installed, however, systems should be implemented such that the number of injection wells as well as the spacing between those injections wells act to maximize RE.

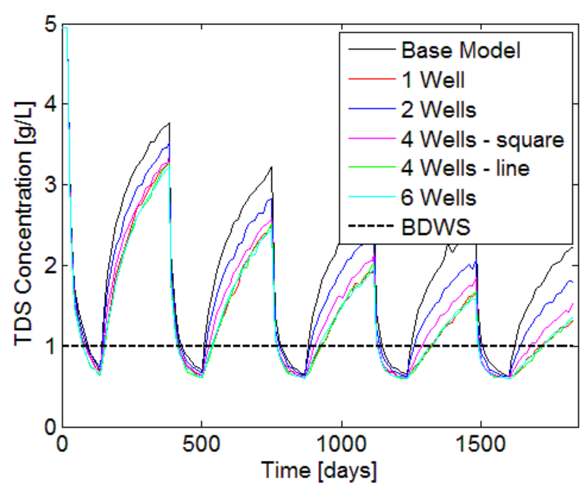

Figure E1: Effect of varying number of injection wells on extraction water TDS concentration.

While the depth of the screening for the extraction well did not significantly impact the extraction water TDS concentrations, the best RE was produced when the extraction well was screened over the entire depth of the aquifer (Figure E2, Table E1). Future site design should consider this alteration to the extraction screen depth rather than just screening in the upper meters of the aquifer.

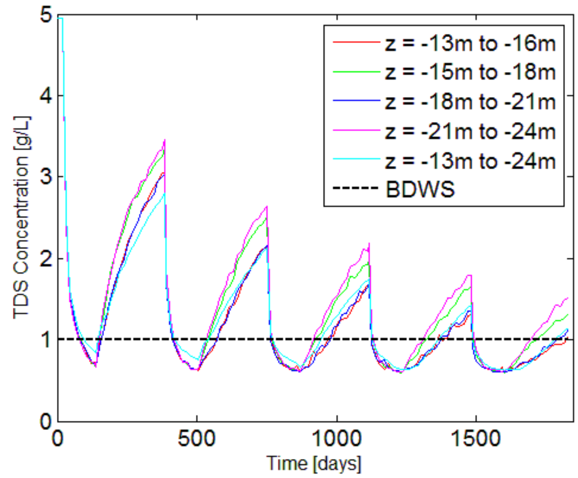

Figure E2: Effect of varying extraction well depth on extraction water TDS concentration.

The extraction and recovery schedule i.e. if extraction took place year-round or extraction only occurred following an injection-only period (daily extraction rate was maintained between models) had a small impact on system RE. Large changes in injection rate, the mass of freshwater in the aquifer, the centroid of the plume and the extent of mixing in the *x* and *z*-directions did not occur between models. Extraction over the full year did improve extraction water TDS concentrations during the first year of injection because of increased drawdown (Figure E3).

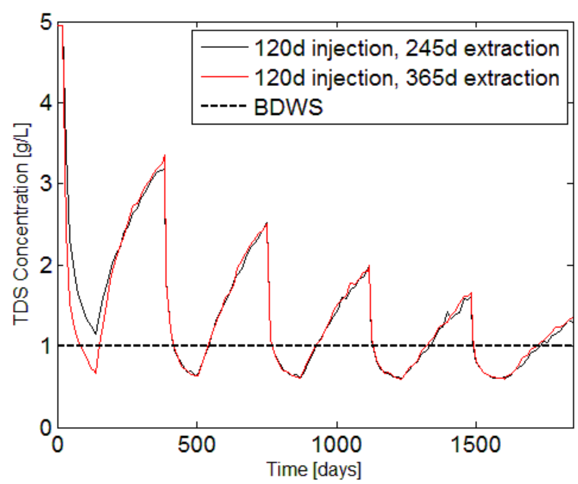

Figure E3: Effect of varying the number of days of extraction on extraction water TDS concentration.

Injection well diameter did affect the injection rate of freshwater into the aquifer and therefore the mass of freshwater in the system and this translated into lower extraction water TDS concentrations. The change from a diameter of 0.1m to 0.5588m was not, however, able to impact the system to a large enough extent to reduce extraction water TDS concentrations to below the BDWS year round (Figure E4). Larger well diameters should be implemented in future system designs to reduce extraction water TDS as much as possible. Mixing in the *x*-direction did increase to a small extent with higher flow rates caused by larger diameters. Mixing in the *z*-direction and shift in the plume radius was minimal.

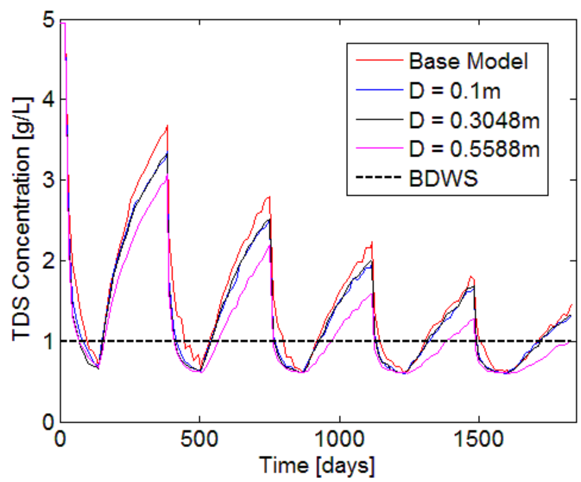

Figure E4: Effect of varying injection well diameter on extraction water TDS concentration.

The relocation of the extraction well to x = 3.065, an additional ~2.5m away from the injection well area, had an influence on extraction water TDS concentrations (Figure E5). The decreased drawdown (enhanced by extraction well proximity) caused lower injection rates and therefore less freshwater mass in the aquifer. This, in addition to the higher TDS concentrations around the extraction well location, served to increase the time in which the system met the BDWS by two injection-recovery cycles. The location of the extraction well had only a small effect on the mixing in the system. In the event that the extraction well were required to be located further from the injection well area, as in the case where greater aquifer retention time is required to attenuate pathogens, a modification to site design should be made so that the extraction well is more centrally located and surrounded by injection wells rather than simply moved further from the injection area.

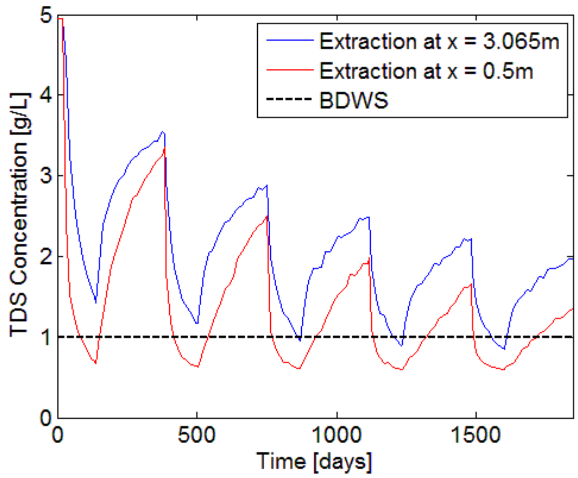

Figure E5: Effect of extraction well location on extraction water TDS concentration.

The Assasuni site has pronounced diurnal hydraulic head fluctuations ($\Delta H_{aquifer} \cong 0.3$m) due to the sites proximity to a tidally-influenced river system. Simulations were performed to examine the influence of the head fluctuations on the system performance. Simulations demonstrated that a diurnal change in head of 0.3m at the injection site was not sufficiently large to impact the system's performance and the extracted water TDS concentrations were similar to when the head fluctuations were not considered (Figure E6).

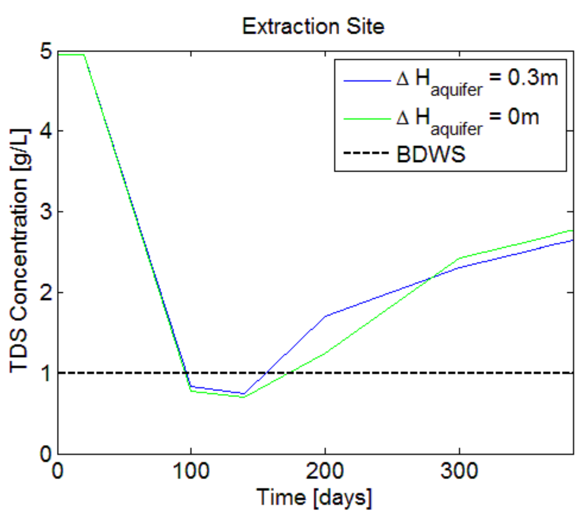

Figure E6: Influence of diurnal aquifer head fluctuations ($\Delta H_{aquifer} \cong$ 0.3m) on extraction well TDS concentration.

Appendix F – Generic Model Set-up

A model of a generic ASTR system configuration (not limited by the existing Assasuni site design) was developed to better examine the factors governing the system efficiency at sites with different hydrogeological and/or design parameters. Following the UN suggested per capita daily water requirement of 20L/d and assuming a 100 person community, a generic model was developed using an extraction rate of 2000 L/d or 2 m^3/d (UN Water, N.D.). According to Ward et al. (2008), in a system with an anisotropy ratio ($K_{x,ave}/K_{z,ave}$) of 2, as adopted by our study, an expected system RE after one cycle would be approximately 52%. Given this RE, the injection rate for the generic case was set to 11.7 m^3/d for water to be extracted at 2 m^3/d at an acceptable TDS concentration for a full 365 days. Injection into the aquifer was specified using an injection well, rather than a general head boundary package as in the Assasuni models. This revision was made to examine the system without the concern of changing flows due to varying injection rates caused by groundwater mounding or non-steady-state conditions. Other parameters of the base generic case are listed in Table F1.

Table F1: Generic base model parameters.

Effective Porosity, n_e [-]	Aquifer Thickness, B [m]	Injection Rate, Q_{inj} [m^3/d]	Extraction Rate, Q_{ext} [m^3/d]	Longitudinal Dispersivity, α_L [m]	Number of Wells	Injection Well Spacing [m]
0.25	11	11.7	2	2.5	4	$L_w^* = 0.25 =$ 1.5m

After determining the required injection and recovery rates, the distance between injection wells and extraction well was then tested for the generic case to determine which well spacing and number of wells could produce the highest performance (performance is based on the ability to maximize RE while still allowing for a minimum retention time of 2.5 days to ensure adequate removal of bacteria from the injectant) (Page et al., 2010). The dimensionless parameter L_w^* was developed to investigate the effect of well spacing and number, where L_w^* is defined as the ratio of the lateral spacing between one injection well and the extraction well (extraction well is located in the centre of the system with injection wells surrounding) to the theoretical plume radius of one injection well:

$$L_w^* = \frac{L_w}{\sqrt{\frac{Q_{net}T}{\pi B n_e}}} \qquad \text{Equation F-1}$$

where Q_{net} is the net injection rate per injection well calculated as $\frac{Q_{inj} - Q_{ext}}{number\ of\ injection\ wells}$ [L^3/T] (note Q_{inj} and Q_{ext} were maintained between models of the same scale) , T is the duration of the injection period [T] and L_w^* is the lateral spacing between one injection well and the extraction well [L]. A summary of the various tested well spacings is given in Table F2.

Table F2: Generic model well spacing test results.

Well Spacing	Distance	Number of Wells	Configuration	RE [%]
L_w^* = 1	6.7 m	3	Triangle	0
L_w^* = 1.5	10.1 m	3	Triangle	0
L_w^* = 0.5	3.4 m	3	Triangle	17.2
L_w^* = 1	6.7 m	4	Rectangle	0
L_w^* = 0.5	3.45 m	4	Rectangle	36.0
L_w^* = 0.*R*	3.4 m	4	Square	44.2
L_w^* = 0.25	1.7 m	4	Square	47.7
L_w^* = 0.25	1.7 m	6	Square	46.9

The RE was highest when four wells was used with L_w^* = 0.25, which produced an RE of 47.7%. The actual system RE is lower than the expected theoretical RE of 52% because the initial start-up phase is not included in the time used to calculate extraction volume (extraction volume = extraction time * extraction rate). The difference between L_w^* = 0.25 spacing and L_w^* = 0.5 spacing with four wells is small, however, L_w^* = 0.25 spacing was chosen for the base generic model as this spacing is more conservative and can adequately meet retention time limits for bacterial removal while reducing land space requirements.

To ensure that the criteria $L_w{}^*$ = 0.25 for well spacing is independent of the size of the ASTR system (size determined by net injection rate), different injection well spacings at different scales were tested (small scale with net injection rate, Q_{net} = 2.4m^3/d, medium scale with net injection rate at Q_{net} = 12.1m^3/d, and large scale with net injection rate at Q_{net} = 24.3m^3/d. The value of R^*_{Disp}, representing the ratio of the theoretical plume size of one injection well to the system longitudinal dispersivity, was maintained between these models at 2.4 to ensure high system efficiency (R^*_{Disp}> 1.5 to ensure systems have RE>0, as determined in Chapter 3). A spacing of $L_w{}^*$ = 0.25 produced the best RE at all scales, yielding an RE of 47.7% (Table F3).

Table F3: Effect of generic model well spacing on system RE.

Model	RE [%]
$L_w{}^*$ = 0.25, small-scale (Q_{net} = 2.4m^3/d)	47.7
$L_w{}^*$ = 0.5, small-scale (Q_{net} = 2.4m^3/d)	44.2
$L_w{}^*$ = 1, small-scale (Q_{net} = 2.4m^3/d)	28.5
$L_w{}^*$ = 0.25, med-scale (Q_{net} = 12.1 m^3/d)	47.7
$L_w{}^*$ = 0.5, med-scale (Q_{net} = 12.1 m^3/d)	46.3
$L_w{}^*$ = 0.25, large scale (Q_{net} = 24.3 m^3/d)	47.7

Some ASTR systems involve a storage period between the injection and recovery phases or a recovery phase which does not overlap with the injection phase. The influence of these different injection-storage-recovery cycles on the system RE and extraction water TDS concentrations was examined and determined to be minimal (Figure F1). A system design using $L_w{}^*$ = 0.25 spacing between injection wells with a centred extraction well can be used effectively with systems independent of the injection-storage-recovery schedule.

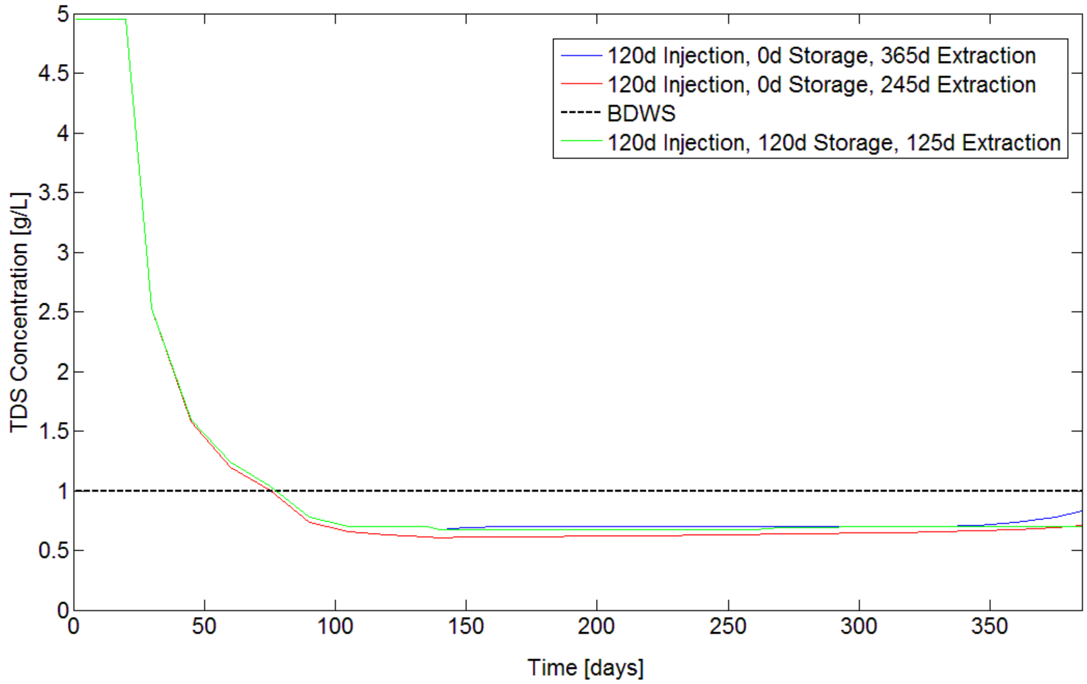

Figure F1: Influence of injection-storage-recovery schedule on extraction TDS concentration.

Simulations were performed to examine the efficiency of systems installed in aquifers with higher ambient groundwater TDS concentrations (10 g/L and 20 g/L). Simulations showed that the BDWS could still be achieved for these higher ambient groundwater concentrations and at an expected RE ~52% (Figure F2). Background TDS concentrations exceeding 20 g/L, however, will cause reduced RE and in this case an alternative system design will be required. Previous studies have shown that macrotilting, caused by density differences between the injected and ambient groundwater, is an issue for systems with high ambient TDS concentrations, especially when a long storage period is used and free convection can dominate (Ward et al., 2009). The ASTR systems simulated in this study do not have a storage period and forced convection between the injection and extraction wells overcomes the free convection that causes macrotilting. As a result, for the conditions simulated ambient groundwater TDS concentrations < 20 g/L do not adversely affect the TDS concentration of the extracted water.

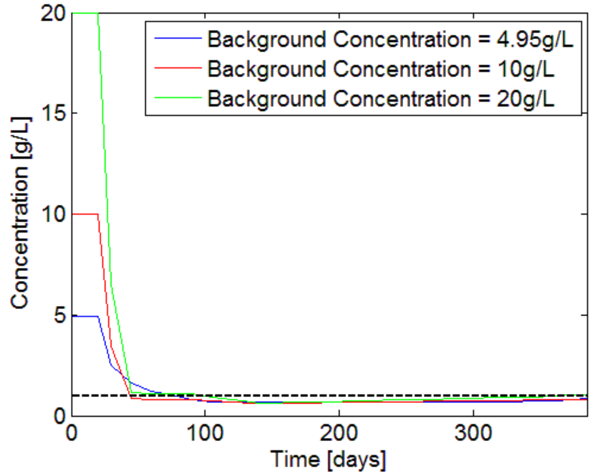

Figure F2: Effect of background TDS concentration on extraction water TDS concentration.

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