Socio-hydrogeological Potential for Managed Aquifer Recharge in the Fresh-saline Aquifers of Southwestern Bangladesh

Floris Loys Naus

Cover picture: Rebecca van Weesep

Chapter pictures: Frank van Broekhoven

Print: Ridderprint | www.ridderprint.nl

Layout: Publiss | www.publiss.nl

SBN: 978-94-6416-185-4

Socio-hydrogeological Potential for Managed Aquifer Recharge in the Fresh-saline Aquifers of Southwestern Bangladesh

Het socio-hydrogeologisch potentieel voor ondergrondse zoetwaterberging in zoet-zout aquifers in Zuidwest Bangladesh

(met een samenvatting in het Nederlands)

Proefschrift

Ter verkrijging van de graad van doctor aan de
Universiteit Utrecht
op gezag van de
rector magnificus, prof. Dr. H.R.B.M. Kummeling,
ingevolge het besluit van het college voor promoties
in het openbaar te verdedigen op

vrijdag 6 november 2020 des ochtends te 11 uur

door

Floris Loys Naus

geboren op 14 februari 1992
te ‘s-Hertogenbosch

Promotoren:

Prof. dr. J. Griffioen

Prof. dr. K.M.U. Ahmed

Copromotor:

Dr. P.P. Schot

The research in this thesis was supported with funding from the Netherlands Organization for Scientific Research (NWO), domain WOTRO Science for Global Development, in the Urbanizing Deltas of the World (UDW) programme, project no. W 07.69.107.

TABLE OF CONTENTS

CHAPTER 1

INTRODUCTION

1. INTRODUCTION

1.1 DRINKING WATER IN COASTAL REGIONS

Worldwide, there are 844 million people that lack a basic drinking water service, and another 263 million people that have limited drinking water services (WHO and UNICEF, 2017). While advances are made to improve the supply of drinking water (UNICEF and WHO, 2019), the current advances are predicted to be not rapid enough to achieve Sustainable Development Goal 6, to 'Ensure availability and sustainable management of water and sanitation for all' (UN, 2015).

One of the most used natural water sources is groundwater. Groundwater is often easily accessible and of good quality. However, proper knowledge of hydrogeology and water resources is required to ensure the safety and sustainability of utilizing groundwater for drinking water (MacDonald and Calow, 2009). Groundwater can be unsafe to drink due to the abundance of natural substances, such as arsenic, fluoride or salinity (van Halem et al., 2009; Ali et al., 2016). Additionally, groundwater can be polluted by surface contaminations from urban waste, agriculture and industry (Pitt et al., 1999; Karim et al., 2013; Leung and Jiao, 2006). Further, groundwater is often unsustainably abstracted, which causes groundwater resources in many parts of the world to become depleted. Clear examples of unsustainable abstraction are present in the upper Ganges, north Arabia, Persia and western Mexico (Gleeson et al. 2012; Wada et al., 2010). Consequential shortages of groundwater may lead to problems with potable water supply and a decrease in produced agricultural goods (Rodell et al., 2009).

In coastal areas, groundwater resources are under pressure from many of these problems. Firstly, with more than half of human population living within 200 km of the coast (Creel, 2003), prominently present intensive agriculture, urbanization and industrialization cause a large anthropogenic stress on groundwater availability. Secondly, coastal areas are sensitive to and expected to be impacted by climate change (Klein and Nicholls, 1999). The often-low elevation at the coast causes both the surface and groundwater to be sensitive to salinization following sea level rise (Ferguson and Gleeson, 2012; Nicholls and Cazenave, 2010; McGranahan et al., 2007; Werner and Simmons, 2009), especially during droughts, which are more likely due to climate change. Therefore, in most coastal areas, the available fresh groundwater is expected to diminish (Ranjan et al., 2006). To ensure the sustainable provision of safe drinking water under these increasing pressures, groundwater resource management is essential.

1.2 MANAGED AQUIFER RECHARGE

An important technology to increase the sustainably available groundwater is Managed Aquifer Recharge (MAR), which involves recharging water to an aquifer with a specific goal (Dillon et al., 2009). There are various goals for which MAR can be applied, including supplying irrigation water, aiding drought-sensitive nature areas and supplying drinking water in both arid and humid regions (Dillon et al., 2009; Dillon, 2005; Maliva et al., 2006). MAR requires a source of water. In areas with a large seasonality in water available from rain or rivers, MAR can be a suitable tool to overcome water stress due to temporal water shortages. MAR can aid with ensuring year-round water supply by capturing and storing water from a period in the year with a large amount of available source water to a period with a small amount of available source water. During storage, the water faces reduced evaporation and water quality deterioration (Page et al., 2018). MAR can also be applied as a step in the treatment of water to improve the water quality. For example, filtration of water through the soil may lead to an improvement of water quality (Dillon et al., 2009). MAR can, therefore, also be a valuable tool during the application of water recycling and reuse practices.

Below, relevant background information on MAR methods (1.2.1), hydrochemical processes during MAR (1.2.2), hydraulic processes during MAR and associated recovery efficiency (1.2.3), and the common problem of clogging (1.2.4) are presented.

1.2.1 MAR types

MAR applications are being implemented worldwide (Figure 1.1) (Stefan and Ansems, 2018). In Hungary, Slovakia, the Netherlands, Germany, Finland, Poland, Switzerland and France, MAR accounts for a significant part of the drinking water supply (Sprenger et al., 2017). There are three main methods to apply MAR (Sprenger et al., 2017; Dillon et al., 2009):

1. *Surface infiltration*. This method involves any technique that lead to enhanced infiltration of water from the surface into the aquifer. Specific examples of this method include spreading water over infiltration basins, infiltrating water through dug wells or trenches, and enhancing infiltration from streams by constructing dams or by removing a low-conductivity streambed. Generally, this method requires the infiltration rate of the topmost layer to be high, but it is also possible to remove a low-conductivity layer at the surface if it is relatively thin. This method

can provide a high recharge rate, can remove turbidity from the water, and can improve quality of the water during filtration through the soil. Therefore, this method is usually applied during the treatment of drinking water.

2. *Well injection.* This method involves utilizing wells to inject water directly into the aquifer, which is required when there is a confining surface layer with a low infiltration rate. MAR applications that use this method are commonly referred to as Aquifer Storage and Recovery (ASR) systems or, when abstraction is at a distance from the recharge well, Aquifer Storage Transfer and Recovery (ASTR) systems. Often, the goal of ASR or ASTR systems is to store water from one period in the year to another.

3. *Bank filtration*. This method does not involve active recharge, but instead involves inducing recharge from a river or lake by pumping nearby groundwater. The lowering of the groundwater table next to the lake or river causes water to flow from the lake or river towards the abstraction well through the aquifer. Most often, this method has as goal to provide drinking water, as water is filtrated during passage of the river or lake bank and the aquifer, leading to a better quality compared to collection of the surface water directly.

In the MAR-portal of IGRAC (Figure 1.1), two other MAR types are recognized: in-channel modification and rainwater and run-off harvesting. In-channel modifications involve enhancing the surface infiltration of water from rivers and streams, and they can, therefore, be seen as a subset of the surface infiltration method (method type 1). Rainwater and run-off harvesting describe a method to collect the source water for MAR, after which MAR can be applied through surface infiltration (method type 1) or well injection (method type 2).

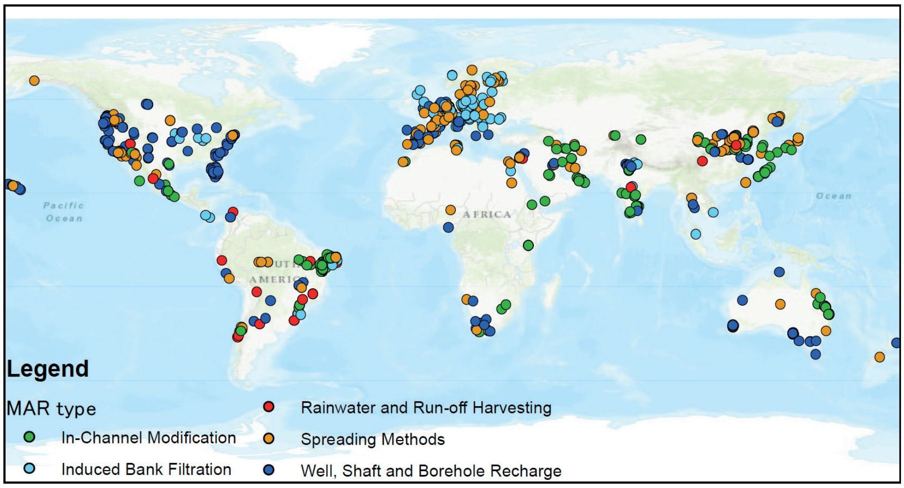


Figure 1.1. Main MAR types throughout the world (Data collected from the IGRAC MAR portal on December 13th, 2019)

1.2.2 Hydrochemical processes during MAR

During passage through the soil and the aquifer, various processes influence the composition of the recharged water. These processes include physical filtering, hydrochemical processes, sorption, and biodegradation. For a large part, these processes attenuate aqueous solutes, improving the water quality. However, they may also lead to deterioration of the water quality. Deterioration can also occur from external sources, such as anthropogenic pollution.

Physical filtering removes suspended solids, including clay particles and colloid-sized pathogenic bacteria and viruses by physically hindering movement through the aquifer (Bradford et al., 2004). Filtering is often paired with physical clogging of the first part of the soil or aquifer.

Hydrochemical processes influence the groundwater composition by processes such as dissolution of minerals and redox transformations. Dissolution of minerals depends on their solubility and on reaction kinetics (Morse and Arvidson, 2002), and can lead to high concentrations of harmful compounds (Ali et al., 2016; Rao et al., 2016; WHO, 2006). Redox processes change the state of compounds following the transfer of electrons, usually catalysed by bacteria in the aquatic environment to kinetically fasten the hydrochemical process (Appelo and Postma, 2005). The redox processes vary depending on the redox state of the environment. Redox processes can result in the mobilization of certain harmful compounds, for example arsenic can be released during pyrite oxidation (Wallis et al., 2010).

Sorption describes the various ways in which a charged or hydrophobic compound can become bound to or taken up into the surface of a solid in the aquifer (Appelo and Postma, 2005). The most relevant charged solids that facilitate sorption consist of clay minerals, oxides and sedimentary organic matter. Sorption largely controls the mobility of trace elements, such as arsenic (Wallis et al., 2010). The mobility varies largely with pH and the composition (including the redox state) of the water (Appelo and Postma, 2005; Calmano et al., 1993). For example, the interplay between redox processes and sorption induces the reductive dissolution of oxides and associated release of sorbed compounds (Pedersen et al. 2006).

Degradation includes all processes where compounds are transformed to other, usually less complex and less harmful compounds, generally by bacteria. Harmful compounds that are found to be degradable or decayable in the aquifer include microorganisms and pathogens (Levantesi et al., 2010), and trace organic chemicals (Rauch-Williams et al., 2010; Alidina et al., 2014). Degradation can occur under various redox conditions, strongly depending on the compound (Rauch-Williams et al., 2010). In MAR, both oxic and reductive degradation is usually facilitated, with oxic degradation occurring in the first part of the aquifer or soil (Hiscock and Grischek, 2002; Grunheid et al., 2005; Harvey et al., 2015), and reductive degradation occurring further downstream in the aquifer (Maeng et al., 2010).

1.2.3 Hydraulic processes during MAR

The performance of MAR systems may be evaluated by the percentage of recharged water that can be recovered with the quality of the water remaining suitable for its use, i.e., the Recovery Efficiency (RE) (Bakker, 2010; Maliva et al., 2006; Ward et al., 2009; Zuurbier et al., 2013; Lowry and Anderson, 2006). When the purpose of MAR is to supply drinking water, the quality of the recovered water should conform to the drinking water standards. Aside from deterioration by processes discussed above, there are also hydraulic processes that can decrease the RE of MAR systems, by causing the recharged water to flow away, making it practically unrecoverable, or causing a mix with native groundwater having insufficient quality to be abstracted, making the water unusable. These hydraulic processes consist of lateral flow, density driven flow, and diffusive-dispersive mixing.

Lateral flow entails the flow of recharged water away from the abstraction well according to the natural hydraulic gradient as present around the MAR system (Ward et al., 2009). This is especially problematic for ASR systems, where any lateral flow will cause the water to flow away from the joint injection-recovery well. ASTR systems are usually designed to account for this background groundwater flow, although design errors or irregular groundwater flow velocities may still be problematic.

Density-driven flow describes the upward flow of recharged water that occurs when there is a density difference between the recharged water and the native groundwater (Bakker, 2010; Ward et al., 2009). This density difference is usually caused by a salinity difference between the recharged water and the native groundwater, which is often the case in coastal areas. The larger the density difference, the larger the buoyant force, and the more likely it is that native groundwater reaches the bottom of the abstraction well. The resulting convection cell will cause recharged water to flow away laterally from the injection/abstraction well, resulting in it becoming unrecoverable (Bakker, 2010; Ward et al., 2009). Density-driven flow is generally irreversible, although MAR design choices can reduce the effects, for example by placing the abstraction well higher than the injection well (Maliva et al., 2006).

Diffusive-dispersive mixing occurs at the fringe between the recharged water and the native groundwater, leading to a mixing zone with reduced concentration gradients. Diffusion occurs due to random movement of particles, which results in mixing whenever there is a concentration gradient (Domenico and Schwartz, 1998). Hydrodynamic dispersion occurs whenever small-scale irregularities, such as aquifer anisotropy and heterogeneity, lead to differences in groundwater flow velocities, for example during lateral flow and density-driven flow (Ward et al., 2009; Domenico and Schwartz, 1998). When the quality of the native groundwater is of sufficient quality, mixing is generally not problematic, and could even lead to improvement of quality due to dilution. However, when the native groundwater is of insufficient quality, these processes can irreversible deteriorate the injected water and can cause abstraction to be halted. In brackish or saline deltas, mixing is problematic because the native groundwater contains too much salinity to be useable.

1.2.4 Clogging

A common problem that affects MAR performance during operation is clogging, which can impair recharge and abstraction rates. Clogging can occur due to a variety of processes (Martin, 2013; Jeong et al., 2018).

Firstly, there is mechanical clogging, caused by colloids or clay minerals getting stuck in smaller pores of the soil or aquifer (Mays, 2013), caused by letting in untreated turbid water. This clogging occurs mostly in the first part of the soil passage. In MAR systems that have as goal to remove turbidity from the water, mechanical clogging is part of the design. In MAR systems that utilize wells, such as ASR and ASTR systems, mechanical clogging is not desirable, as it is hard to rehabilitate clogged wells. For such systems, the water requires to be filtered prior to injection to remove excess turbidity.

Secondly, there is chemical clogging, consisting of the precipitation of minerals and amorphous solids (Martin, 2013). The most common form of chemical clogging occurs when reduced water mixes with oxidized water, inducing the precipitation of iron hydroxides (Martin, 2013). Chemical clogging can also occur under reduced conditions, for example in the form of iron sulphide precipitation under sulphate reducing conditions (van Beek, 1984). Precipitation of minerals linked to pH changes can also occur, for example calcium carbonate precipitation (Martin, 2013; van Beek, 1984).

Thirdly, there is biological clogging, caused by the growth of a biofilm under favourable conditions, for example, when the inlet water contains high substrate concentrations, such as organic matter (Battin and Sengschmitt, 1999; Kim et al., 2010). It should be noted that, while biological activity can decrease the recharge rate, it is often beneficial for the water quality, as described above in section 1.2.2.

There are various techniques to rehabilitate a MAR system affected by clogging. For MAR systems with infiltration from surface basins, the clogged top layer can be easily reached, so the most straightforward treatment is to remove this layer. For clogged recharge or abstraction wells, rehabilitation is less straightforward. Nevertheless, there are several methods to rehabilitate a clogged well. The well filter can be cleaned directly using a brush or using pressure cleaning. Alternatively, the clogged material can be removed by various methods that increase the water flow, for example by increased pumping from the well, or by invoking high flow rates by placing the filter under high pressure using air. Further, there are chemical methods to rehabilitate a clogged well. Most commonly, acids are used, as they can remove various compounds, for example calcite, hydroxides or biomass (van Beek, 1984; De la Loma Gonzalez, 2013). Oxidisers and reductants can also be used to remove precipitated reduced or oxidised material, respectively.

1.3 COASTAL ZONE OF SOUTHWESTERN BANGLADESH

The southwestern coastal zone of Bangladesh is a salinizing coastal region that may potentially benefit from the application of the MAR technology to overcome the currently problematic provision of safe drinking water (section 1.3.1). The region is a subpart of the Ganges-Brahmaputra-Meghna (GBM) delta, which has been described to be under extreme pressure from a social and resources management point-of-view (e.g. Sharma et al., 2010).

The region consists of the districts Satkhira, Khulna and Bagerhat (Figure 1.2) and is characterized by low-lying polders protected by dikes that have been constructed in the 1960s to decrease the flooding risk. The polders are subsiding due to resettling of the sediments and a decrease in sediment supply to the land after the polders were created (Auerbach et al., 2015). This lack of sedimentation, and the consequential subsidizing, is clearest when comparing the polders with the Sundarbans, the largest mangrove forest in the world, located in the south of the region (Figure 1.2). The Sundarbans are a protected forest without permanent settlement and without flood protection, and as a consequence, this forest had continued sedimentation, and now has a higher elevation than the low-lying polders more inland (Auerbach et al., 2015).

Rainfall in the region is around 2500 mm a year, with most rain falling in the monsoon season from July to October. The monsoon season is followed by a dry winter from October-March and a hot summer from March-June. There are many tidal rivers throughout the region that are fresh in the wet season but become saline in the dry season (Bhuiyan and Dutta, 2012). The saline water influence on the tidal rivers is expected to increase under climate change scenarios (Zaman et al., 2017), and seawater flooding is expected to increase due to an increase in tropical storms with associated storm surges (Knutson et al., 2010).

The region is predominantly rural, with about 80% of the 5.8 million people living in rural conditions (Bangladesh Bureau of Statistics, 2011). People mostly live in villages with trees, located in-between the agricultural fields and aquaculture ponds, usually on slightly higher elevated land than the surrounding areas. The livelihoods of the people are vulnerable to multiple stressors that may impact food and water security (Shameem et al., 2014). Borgomeo et al. (2017) speaks of a poverty trap, due to the interaction of flooding, salinity, agricultural gains and income. The main forms of income are agriculture, mostly in the form of rice farming, and aquaculture, mostly in the form of fish and shrimp farming. In the north, irrigation enables multiple agricultural growth seasons, but in the part of the area close to the Sundarbans, a lack of available groundwater causes agriculture to only be possible during the wet season. Instead, aquaculture is prominent, resulting in large parts of the land being permanently inundated. For most forms of aquaculture, such as shrimp farming, brackish or saline water is required. The aquaculture has been linked to various social and environmental problems, such as social unrest, mangrove degradation, saltwater intrusion, and surface water and soil pollution (Paul and Vogl, 2011; Azad et al., 2009).

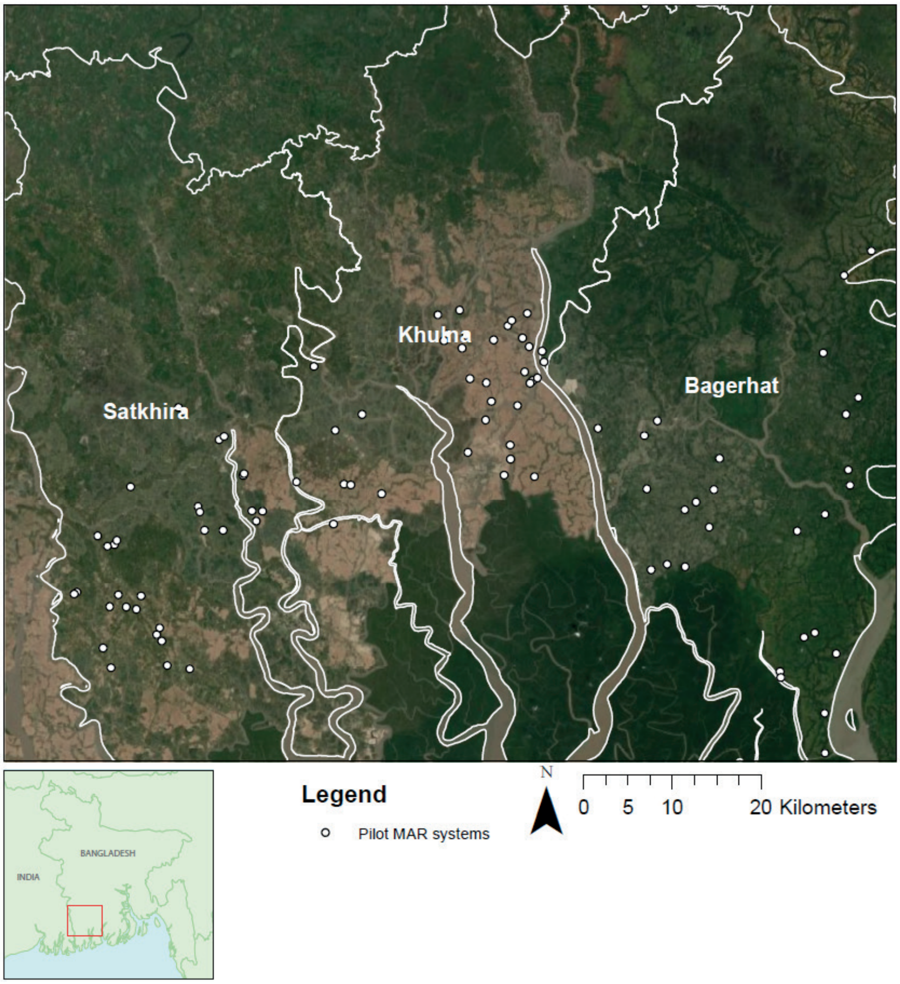


Figure 1.2. Pilot MAR sites in the study region consisting of the southwestern Bangladesh coastal districts Satkhira, Khulna and Bagerhat.

1.3.1 Drinking water provision

The drinking water provision in the southwestern region of Bangladesh is characterized by a variety of water sources, with several problems in terms of either availability or quality.

Rainwater is commonly used for drinking in the wet season, when abundant rainfall occurs (Chowdhury, 2010), but for usage during the dry season rainwater needs to be stored. Traditionally, rainwater is stored in ponds, and there is a long history of protection of designated ponds for drinking water (Kräzlin, 2000). Nevertheless, the ponds are often contaminated by bacteria, leading to acute health problems (Islam et al., 2011; Alam et al 2006; Knappett et al., 2011). Besides storage in ponds, the rainwater is nowadays also stored in large tanks, but the quality of the water is found to still deteriorate during long term storage (Islam et al., 2010; Islam et al., 2011; Simmons et al., 2001; Despins et al., 2009; Dobrowsky et al., 2014).

Since the 1960s, groundwater has been introduced as a drinking water source to reduce the health risks associated with the consumption of pond water. In the 1990s, it became known that much of the shallow groundwater in the GBM delta has a high arsenic concentration (Nickson et al., 1998; Harvey et al., 2002; BGS and DPHE 2001; Fendorf et al., 2010). Additionally, the groundwater in southwestern Bangladesh is found to regularly contain high salinity levels (George, 2013; Worland et al., 2015; Ayers et al., 2016; Rahman et al., 2018). Consequently, groundwater often exceeds the Bangladesh drinking water standards for arsenic and salinity (EC is 2 mS/cm, As is 0.05 mg/L; Ayers et al., 2016). Currently, groundwater is used for both drinking water and irrigation, leading to unsustainable extraction rates compared to the low recharge rate (Shamsudduha et al., 2011).

The occurrence of water shortages and water quality problems has resulted in the emergence of water vendors, but the source and quality of vendor water is not always clear (Kjellén and McGranahan, 2006). Additionally, the vendor water is not affordable for all the people living in the region. Recently, both public and private investments in deep tubewells and the introduction of other technologies to supply drinking water in the region are increasing (Hoque et al., 2019). Various technical solutions have been implemented in the region. For example, Pond Sand Filters (PSF) have been constructed to improve the quality of the pond water for drinking water purposes (Yokota et al., 2001), but they do not manage to completely remove the microbiological contamination (Harun and Kabir, 2013; Islam et al., 2011). Additionally, Reverse Osmosis (RO) systems have been implemented, which can remove bacteria, salinity and arsenic from water, although they do not always successfully lower salinity in some occasions (Islam et al., 2018; Ning 2002), and the produced water require the addition of minerals. Another relatively recent development is the construction of pipeline systems, which distribute good quality deep, but fossil, groundwater to areas with limitations in terms of availability or quality. Recently, MAR systems have been piloted as a sustainable alternative water provision option, which will be discussed in more detail in section 1.4.

The implementation of safe technical solutions does not guarantee that people adopt and use them. For example, Kabir and Howard (2007) found that only 36% of the installed arsenic-free drinking water options in Bangladesh were functional. Likewise, many of the installed PSFs have been abandoned, as evident by the non-functional PSFs throughout the villages. People are attached to their current water options, so a behavioural change is required for people to adopt safe drinking water options (Hurlimann et al. 2016; Du Preez et al. 2010; Mäusezahl et al., 2009).

1.3.2 Complex hydrogeology and groundwater salinity

In southwestern Bangladesh, the hydrogeology and groundwater salinity are characterized by a larger variation, which is poorly understood due to a lack of research and data (Ayers et al., 2016; Mukherjee et al., 2009). Generally, three groups of aquifers are recognized (Mukherjee et al., 2009). The shallowest aquifers are the Upper Holocene aquifers, which extend down to a depth of approximately 60 m in the south. Below, the Middle Holocene aquifers are located, followed by the Late Pleistocene-Early Holocene aquifers. The depth of the interface between the Middle Holocene and the Late Pleistocene-Early Holocene aquifers varies. The divide between the Pleistocene and the Holocene is characterized by the landscape during the Last Glacial Maximum (LGM), when river erosion led to deeply incised valleys down to approximately 120 m deep compared to the present-day surface (BGS and DPHE, 2001; Hoque et al., 2014; Mukherjee et al., 2009). The palaeo interfluve areas between the valleys are less deep and are characterized by a palaeosol consisting of oxidized sands (Umitsu, 1993; Burgess et al., 2010; Hoque et al., 2014). The Holocene aquifers are formed during the transgression and progression, with the transgression being characterized by rapid sedimentation and a receding coastline (Goodbred and Kuehl, 2000), and the progression being characterized by an out-building coastline (Goodbred and Kuehl, 2000; Goodbred et al., 2014). During the last sedimentary period in the Holocene, the Ganges migrated eastwards away from southwestern Bangladesh, resulting in the deposition of the covering clay layer. A variety of soils are present in the region, with fluvial soils in the north of the region, tidal fringe soils in the east of the region, and tidal flat soils in the south of the region (FAO, 1959).

The groundwater of southwestern Bangladesh is characterized by low groundwater velocities and the presence of both arsenic and salinity (Ayers et al., 2016; Bahar and Reza, 2010; Nickson et al., 1998; Worland et al. 2015). Arsenic is generally considered to originate from hydroxides in the sediments transported to Bangladesh by the large rivers during the Holocene, where subsequent reduction of organic matter results in the release of arsenic from the hydroxides (Nickson et al., 1998; Harvey et al. 2002; Ravenscroft et al., 2005; Winkel et al., 2008). The organic matter was originally linked to pond water influence (Harvey et al., 2002; Harvey et al., 2006), but it is unlikely that this is the sole source because infiltration from ponds is very limited (Sengupta et al., 2008). Arsenic occurs more often and in higher concentrations in the relatively shallow groundwater, down to a depth of approximately 100 meters, while deeper groundwater generally has a lower arsenic concentration (BGS and DPHE 2001; Fendorf et al., 2010). In the relatively shallow groundwater (< 100 m), low arsenic concentrations are described to be found under two circumstances (Hoque et al., 2011). Firstly, arsenic is found to sorb to the oxidised sands of the buried palaeosol present, causing groundwater in and below the palaeosol to generally have a low arsenic concentration (Hoque et al., 2014). Secondly, arsenic concentrations are found to be low in very shallow areas with a high conductive soil, where arsenic is flushed away or where more oxic conditions occur (Radloff et al., 2017; Ravenscroft et al., 2005; Weinman et al., 2008).

Salinity of the groundwater in southwestern Bangladesh is described to vary largely (Ayers et al., 2016; George, 2013; Worland et al., 2015). Parts of the groundwater are described to consist of connate groundwater, with the salinity being determined by the moment at which the aquifers became sealed off (Ayers et al., 2016; Worland et al., 2015). Some limited present-day processes have been proposed to occur in areas with a relatively thin clayey top layer, such as infiltration of saline water from aquaculture ponds (Paul et al., 2011), evaporite salts redissolving in the soil during the wet season (Sarker et al., 2018) and infiltration of brackish or saline water from tidal rivers (Ayers et al., 2016; Rahman et al., 2000), as well as infiltration of freshwater from rain or freshwater ponds (Ayers et al., 2016). However, much salinity variation had not been explained, and the exact controlling processes were unclear.

1.4 MAR IN SOUTHWESTERN BANGLADESH

1.4.1 UNICEF MAR pilot

In southwestern Bangladesh, MAR has been piloted as a suitable sustainable drinking water option in parts of the region where none of the other safe alternatives are available in the form of small-scale, community-run systems (Figure 1.3) in a project funded by UNICEF. These MAR systems are akin to ASTR systems (method type 2, section 1.2.1) in functionality, but they are locally referred to as MAR systems (Sultana et al., 2014). This project started with a pilot of 20 systems from 2009 to 2013, followed by the construction of another 79 pilot MAR systems in 2013 and 2014. These MAR systems store water from the monsoon period in the subsurface for use during the dry season (Figure 1.3). During storage, the water quality in terms of bacterial contamination is expected to improve (see section 1.2.2). The source water for the MAR systems is from surface ponds, so to prevent mechanical clogging (see section 1.2.4) water is first filtered through a sand filter, and then infiltrated by 4 to 6 infiltration wells around an abstraction well. The infiltration wells have a filter length of 9.1 m (30 feet) situated around a single abstraction well with a filter length of 3 m (10 feet) long that is installed approximately 3 m (10 feet) higher in the aquifer (Acacia, 2014). Contrary to the distinct seasonality in precipitation, the MAR systems do not have separate injection and abstraction periods, i.e. injection and abstraction occur more or less continuously throughout the year, although on average, recharge peaks in the wet season and abstraction peaks in the water-stressed dry season leading to overall storage from the wet season to the dry season. The simultaneous injection and abstraction may lead to part of the water having a relatively low storage time, and consequently have less time for microorganisms to die-off (Sultana et al., 2014). After installation, the MAR systems were donated to the local communities, which were expected to run and maintain the systems themselves.

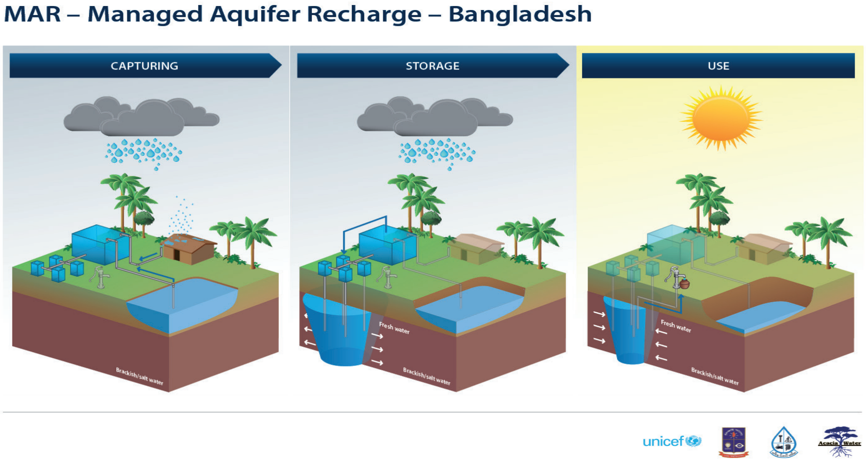


Figure 1.3. Schematic overview of the pilot MAR systems (Acacia, 2014).

1.4.2 Delta-MAR research project

Both success and failure were observed in the pilot MAR systems. Some MAR systems did not manage to produce water of sufficient quality, indicating unforeseen hydrochemical and hydraulic processes. Other MAR systems were not well-operated or maintained, indicating problems with the governance structure of the MAR systems. Additionally, much was unknown about whether MAR could be successful throughout the southwestern region in Bangladesh, related to a lack of understanding of the hydrogeological conditions throughout the region, as described in section 1.3.2.

The Delta-MAR scientific research project was started to address the above-mentioned problems and knowledge gaps. It was funded by the domain 'NWO-WOTRO Science for Global Development' of the Dutch Research Council, in the Urbanizing Deltas of the World (UDW) programme (Grant number: OND1357179). It was undertaken by a consortium of Utrecht University, Delft University of Technology, Dhaka University and Acacia Water. The Delta-MAR research was set-up around four PhD research projects. The first 3 projects were in a so-called sandwich construction with shared affiliation between Dhaka University and either Utrecht University or Delft University of Technology, and aimed to elucidate knowledge gaps regarding 1) hydraulic processes during MAR, and effects of design and operation choices; 2) effects of hydrochemical processes on the water quality during MAR; and 3) effects of the governance structure on the success of MAR systems. This thesis describes the research of the fourth PhD project, which focusses on 4) the potential of MAR systems being successful in the hydrogeologically highly variable and data-poor region of southwestern Bangladesh.

1.5 THIS THESIS: POTENTIAL FOR MAR IN SOUTHWESTERN BANGLADESH

The potential of MAR systems being successful in southwestern Bangladesh depends on two main aspects, based on the two common observed pitfalls of the pilot MAR systems. Firstly, the MAR systems require suitable hydrogeological conditions to technically function well, i.e., produce enough water of acceptable quality. Secondly, their success also depends on whether people need them, and would want to adopt, operate and use them, since these systems were implemented in a supply-driven manner and were aimed to be community-run.

The suitability of areas for technically well-functioning MAR systems has been researched in various regions in the world, mostly with a focus on systems that utilize infiltration basins, and usually by expert-judgement (e.g. Rahman et al., 2012; Ghayoumian et al., 2007; Kallali et al., 2007). In southwestern Bangladesh, the MAR systems are required to use injection wells (Figure 1.3), because the subsurface is characterized by a clayey top layer (Mukherjee et al., 2009). Thus, their performance is mostly determined by subsurface processes and conditions, which vary according to the hydrogeology (Ward et al., 2009; Bakker et al., 2010), as discussed in section 1.2.3. To determine the suitability of the hydrogeology for MAR, sufficient data on and understanding of the lithological situation and of the groundwater salinity are required. However, as discussed in section 1.3.2, southwestern Bangladesh is a relatively data-poor region, especially in relation to the large variation in lithology of the unconsolidated sediments. As a consequence, there is no clear overview of both the lithological situation and the groundwater salinity in the region.

There have been various initiatives and programs that focussed on improving methods to assess or predict hydrological processes and responses in data-poor regions. For example, the program on Predictions in Ungauged Basins (PUB) (Hrachowitz et al., 2013), that focussed on improving hydrological predictions in ungauged basins by improving scientific understanding of underlying hydrological processes and by improving and developing predictive models. Additionally, there has been the more problem-oriented Challenge Programme on Water and Food, Basin Focal Projects (Mulligan et al., 2011), that had as goal to address hydrological problems in data-poor basins to improve water management and food production. Various approaches have been used in the studies on data-poor or ungauged basins. There have been studies that focussed on direct assessment, for example by applying remote sensing imagery as a proxy for surface hydrology (e.g. Thieme et al., 2007) or for water resources assessments (e.g. Liebe et al., 2009).

Other studies have focused on applying knowledge on and understanding of processes and models from data-rich environments towards data-poor or ungauged basins, for example, the comparative hydrology approach (e.g. Singh et al., 2014). Lastly, there have studies that focussed on developing large scale hydrological water balance models that can be applied in data-poor regions (e.g. Mulligan, 2013). However, previous studies in data-poor regions or ungauged basins have mostly focused on problems related to streamflow, floods, water quality, water supply, and sustainability of water resources (Hrachowitz et al., 2013; Mulligan et al., 2011). There has been no focus on elucidating or predicting the groundwater salinity variation in areas with a lack of data, or on the development of methods to link underlying processes that steer the hydrogeology and groundwater salinity to proxies available at a larger scale. Therefore, to assess the suitability of areas for MAR in southwestern Bangladesh, methods for obtaining an overview of the hydrogeological conditions, in specific the groundwater salinity, are required.

In published studies on MAR potential (e.g. Rahman et al., 2012; Ghayoumian et al., 2007; Kallali et al., 2007; Zuurbier et al., 2013), the social necessity and acceptance for MAR was inherently assumed to be present. However, it is not obvious to assume that this is the case for MAR in the southwestern region of Bangladesh. This is especially important, as the systems are aimed to be community-run after they are adopted by the local communities. For people to adopt the MAR systems, a behavioural change is needed, and the likelihood of such a change occurring varies per water option (Hurlimann et al. 2016; Du Preez et al. 2010; Mäusezahl et al., 2009). Therefore, it is important to elucidate what controls the attachment of the local communities to their current water options to be able to assess whether and where MAR might be successfully adopted when they would be implemented.

1.6 RESEARCH AIM AND OUTLINE THESIS

The general aim of this thesis is to understand the potential of MAR systems being successful in the complex hydrogeological setting of southwestern Bangladesh. This potential is approached from a hydrogeological and a social point-of-view (Figure 1.4). A large part of the study focusses on elucidating the previously largely unknown hydrogeological situation in the region, in specific related to the variation in groundwater salinity. The hydrogeological situation determines whether an area will facilitate a technically well-performing MAR system. Also, the hydrogeological situation is related to the necessity for MAR, as it determines whether groundwater can safely be used for drinking. To be able to elucidate the social potential for MAR, this thesis additionally focusses on the attachment of people to their unsafe water options in the region, as this influences their aptitude to accept MAR. Lastly, regional differences in necessity and technical suitability for MAR are combined, to elucidate the socio-hydrogeological potential for MAR needed to assess and improve future MAR implementation in the region.

The expectation is that the findings will aid 1) to create an overview as well as a management strategy of the current water resources in the region, 2) to create a strategy for MAR implementation and upscaling throughout the region and 3) to sustainably solve the drinking water problems in the region.

The thesis has the following specific aims (Figure 1.4):

- Develop a conceptual model of the spatio-temporal groundwater salinity evolution through the Holocene to understand the spatial groundwater salinity variation as a function of the controlling palaeo and present-day processes (Chapter 2)
- Understand and link these controlling processes to surface characteristics and groundwater salinity at a regional scale (Chapter 3)
- Assess to what extent and why users are attached to their current, unsafe water options (Chapter 4)
- Determine the regional potential for MAR based on regional differences in necessity and technical suitability (Chapter 5)

The general methods of the study involve local and regional hydrogeological fieldwork, conducting interviews, analysis of water samples in the laboratory, PHREEQC modelling, statistical analysis and GIS analysis.

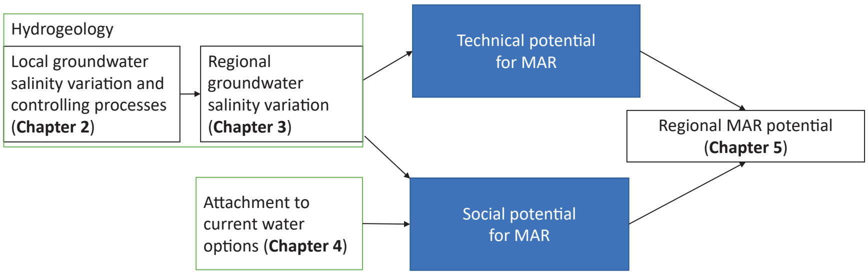


Figure 1.4. Schematic overview of the research presented in this thesis.

In chapter 2, high-density hydrogeological research in a local study area is presented. The fieldwork involved collecting data on lithology, as well as the chemical and isotopic groundwater composition. With the application of a hydrogeological palaeo reconstruction from literature, reactive transport modelling using PHREEQC, and analysis of isotopic and chemical samples, we developed a conceptual model of the groundwater salinity evolution throughout the Holocene. This provides a framework of the relevant palaeo and present-day hydrogeological processes that control groundwater salinity, which can be expected throughout the region.

In chapter 3, the regional groundwater salinity variation is researched. Based on a low-cost method of measuring electrical conductivity values of groundwater from tube wells, an overview of the regional groundwater salinity was constructed. Using statistical analyses and a geomorphological analysis, the extent to which surface characteristics can be linked to hydrogeological processes and groundwater salinity was determined. Lastly, rules are constructed to predict the groundwater salinity throughout the southwestern region of Bangladesh based on surface characteristics.

In chapter 4, the behaviour surrounding the usage of safe and unsafe water options is analysed. A survey study was conducted to determine how attached people are to their current water option and to determine possible behavioural facilitators and barriers of the usage of unsafe drinking water options. Lastly, implementation strategies were discussed to ensure successful adoption of new alternative technologies that provide safe drinking water.

In chapter 5, regional differences in potential for MAR are researched. A regional groundwater quality database, a regional lithological database, and land cover data were used to assess 1) the varying necessity for MAR throughout the region, and 2) the varying technical suitability of areas for MAR throughout the region based on constraints for MAR and the expected effect of density-driven flow and vulnerability of the MAR systems. By combining these two assessments, an overview of the varying regional opportunities for MAR was provided, which is useful for a first step in future site-selection for MAR in the region and for identifying strategies to increase the potential that MAR will be successful.

Finally in the synthesis, chapter 6, the findings of the research are summarized and evaluated in relation to MAR site-selection, implementation and operation. Additionally, the relevance of the findings for other deltas and coastal regions across the world are discussed. Lastly, the thesis gives recommendations for policy makers in the region and for further research.

REFERENCES

Acacia Water, 2014. Manual for MAR system implementation in saline areas Report No. 6, Acacia Water, Gouda

Alam, M., Sultana, M., Nair, G.B., Sack, R.B., Sack, D. a, Siddique, a K., Ali, A., Huq, A., Colwell, R.R., 2006. Toxigenic Vibrio cholerae in the Aquatic Environment of Mathbaria, Bangladesh. Appl. Environ. Microbiol. 72, 2849–2855. https://doi.org/10.1128/AEM.72.4.2849-2855.2006

Ali, S., Thakur, S.K., Sarkar, A., Shekhar, S., 2016. Worldwide contamination of water by fluoride. Environ. Chem. Lett. 14, 291–315. https://doi.org/10.1007/s10311-016-0563-5

Alidina, M., Li, D., Drewes, J.E., 2014. Investigating the role for adaptation of the microbial community to transform trace organic chemicals during managed aquifer recharge. Water Res. 56, 172–180. https://doi.org/10.1016/j.watres.2014.02.046

Appelo, C.A.J., Postma, D., 2005. Geochemistry, groundwater and pollution, 2nd ed. A.A. Balkema, Rotterdam.

Auerbach, L.W., Goodbred Jr, S.L., Mondal, D.R., Wilson, C. a., Ahmed, K.R., Roy, K., Steckler, M.S., Small, C., Gilligan, J.M., Ackerly, B. a., 2015. Flood risk of natural and embanked landscapes on the Ganges–Brahmaputra tidal delta plain. Nat. Clim. Chang. 5, 153–157. https://doi.org/10.1038/nclimate2472

Ayers, J.C., Goodbred, S., George, G., Fry, D., Benneyworth, L., Hornberger, G., Roy, K., Karim, M.R., Akter, F., 2016. Sources of salinity and arsenic in groundwater in southwest Bangladesh. Geochem. Trans. 17, 4. https://doi.org/10.1186/s12932-016-0036-6

Azad, A.K., Jensen, K.R., Lin, C.K., 2009. Coastal aquaculture development in Bangladesh: Unsustainable and sustainable experiences. Environ. Manage. 44, 800–809. https://doi.org/10.1007/s00267-009-9356-y

Bahar, M.M., Reza, M.S., 2010. Hydrochemical characteristics and quality assessment of shallow groundwater in a coastal area of Southwest Bangladesh. Environ. Earth Sci. 61, 1065–1073. https://doi.org/10.1007/s12665-009-0427-4

Bakker, M., 2010. Radial Dupuit interface flow to assess the aquifer storage and recovery potential of saltwater aquifers. Hydrogeol. J. 18, 107–115. https://doi.org/10.1007/s10040-009-0508-1

Bangladesh Bureau of statistics, 2011. Population and Housing Census 2011 - National report, Volume 4. ed. Dhaka.

Battin, T.J., Sengschmitt, D., 1999. Linking sediment biofilms, hydrodynamics, and river bed clogging: Evidence from a large river. Microb. Ecol. 37, 185–196. https://doi.org/10.1007/s002489900142

BGS, DPHE, 2001. Arsenic contamination of groundwater in Bangladesh; BGS Technical Report WC/00/19. British Geological Survey, Keyworth.

Bhuiyan, M.J.A.N., Dutta, D., 2012. Assessing impacts of sea level rise on river salinity in the Gorai river network, Bangladesh. Estuar. Coast. Shelf Sci. 96, 219–227. https://doi.org/10.1016/j.ecss.2011.11.005

Borgomeo, E., Hall, J.W., Salehin, M., 2018. Avoiding the water-poverty trap: insights from a conceptual human-water dynamical model for coastal Bangladesh. Int. J.

Water Resour. Dev. 34, 900–922. https://doi.org/10.1080/07900627.2017.1331842

Bradford, S.A., Bettahar, M., Simunek, J., van Genuchten, M.T., 2004. Straining and attachment of colloids in physically heterogeneous porous media. Vadose Zo. J. 3, 384–394. https://doi.org/10.2113/3.2.384

Burgess, W.G., Hoque, M.A., Michael, H.A., Voss, C.I., Breit, G.N., Ahmed, K.M., 2010. Vulnerability of deep groundwater in the Bengal Aquifer System to contamination by arsenic. Nat. Geosci. 3, 83–87. https://doi.org/10.1038/ngeo750

Calmano, W., Hong, J., Forstner, U., 1993. BINDING AND MOBILIZATION OF SEDIMENTS AFFECTED BY pH AND REDOX POTENTIAL. Wal. Sci. Tech 28, 223–235.

Chowdhury, N.T., 2010. Water management in Bangladesh: an analytical review. Water Policy 12, 32. https://doi.org/10.2166/wp.2009.112

Creel, L., 2003. Ripple effects: Population and coastal regions, Population Reference Bureau.

De La Loma Gonzalez, B., 2013. Clogging of deep well infiltration recharge systems in the Netherlands. Clogging issues associated with managed aquifer recharge methods In: Martin R (ed.) Clogging issues associated with managed aquifer recharge methods. IAH Commission on Managing Aquifer Recharge, Australia, 163–173.

Despins, C., Farahbakhsh, K., Leidl, C., 2009. Assessment of rainwater quality from rainwater harvesting systems in Ontario, Canada. J. Water Supply Res. Technol. - AQUA 58, 117–134. https://doi.org/10.2166/aqua.2009.013

Dillon, P., 2005. Future management of aquifer recharge. Hydrogeol. J. 13, 313–316. https://doi.org/10.1007/s10040-004-0413-6

Dillon, P., Pavelic, P., Page, D., Beringen, H., Ward, J., 2009. Managed aquifer recharge: An Introduction, Waterlines Report Series.

Dobrowsky, P.H., De Kwaadsteniet, M., Cloete, T.E., Khan, W., 2014. Distribution of Indigenous Bacterial Pathogens and Potential Pathogens Associated with Roof-Harvested Rainwater. Appl. Environ. Microbiol. 80, 2307–2316. https://doi.org/10.1128/aem.04130-13

Domenico, P.A., Schwartz, F.W., 1998. Physical and chemical hydrogeology. Second edition. Phys. Chem. Hydrogeol. Second Ed.

Du Preez, M., McGuigan, K.G., Conroy, R.M., 2010. Solar disinfection of drinking water in the prevention of dysentery in South African children aged under 5 years: The role of participant motivation. Environ. Sci. Technol. 44, 8744–8749. https://doi.org/10.1021/es103328j

FAO, 1959. Soil Survey of the Ganges-Kobadak Area. ETAP Report no. 1071. Rome.

Fendorf, S., Michael, H.A., Geen, A. Van, 2010. Spatial and Temporal Variations of Groundwater Arsenic in South and Southeast Asia — Supporting Online Material 1123–1128.

Ferguson, G., Gleeson, T., 2012. Vulnerability of coastal aquifers to groundwater use and climate change. Nat. Clim. Chang. 2, 342–345. https://doi.org/10.1038/nclimate1413

George, G.J., 2013. Characterization of salinity sources in southwestern Bangladesh evaluated through surface water and groundwater geochemical analyses. Vanderbilt University.

Ghayoumian, J., Mohseni Saravi, M., Feiznia, S., Nouri, B., Malekian, A., 2007. Application of GIS techniques to determine areas most suitable for artificial groundwater recharge in a coastal aquifer in southern Iran. J. Asian Earth Sci. 30, 364–374. https://doi.org/10.1016/j.jseaes.2006.11.002

Gleeson, T., Wada, Y., Bierkens, M.F.P., Van Beek, L.P.H., 2012. Water balance of global aquifers revealed by groundwater footprint. Nature 488, 197–200. https://doi.org/10.1038/nature11295

Goodbred, S.L., Kuehl, S.A., 2000. Enormous Ganges-Brahmaputra sediment discharge during strengthened early Holocene monsoon. Geology 28, 1083. https://doi.org/10.1130/0091-7613(2000)28<1083:EGSDDS>2.0.CO;2

Goodbred, S.L., Paolo, P.M., Ullah, M.S., Pate, R.D., Khan, S.R., Kuehl, S.A., Singh, S.K., Rahaman, W., 2014. Piecing together the Ganges-Brahmaputra-Meghna River delta: Use of sediment provenance to reconstruct the history and interaction of multiple fluvial systems during Holocene delta evolution. Geol. Soc. Am. Bull. 126, 1495–1510. https://doi.org/10.1130/B30965.1

Grünheid, S., Amy, G., Jekel, M., 2005. Removal of bulk dissolved organic carbon (DOC) and trace organic compounds by bank filtration and artificial recharge. Water Res. 39, 3219–3228. https://doi.org/10.1016/j.watres.2005.05.030

Harun, M. a. Y. a., Kabir, G.M.M., 2012. Evaluating pond sand filter as sustainable drinking water supplier in the Southwest coastal region of Bangladesh. Appl. Water Sci. 3, 161–166. https://doi.org/10.1007/s13201-012-0069-7

Harvey, C.F., Swartz, C.H., Badruzzaman, A.B.M., Keon-Blute, N., Yu, W., Ali, M.A., Jay, J., Beckie, R., Niedan, V., Brabander, D., Oates, P.M., Ashfaque, K.N., Islam, S., Hemond, H.F., Ahmed, M.F., 2002. Arsenic Mobility and Groundwater Extraction in Bangladesh. Science (80-.). 298, 1602–1606. https://doi.org/10.1126/science.1076978

Harvey, C.F., Ashfaque, K.N., Yu, W., Badruzzaman, A.B.M., Ali, M.A., Oates, P.M., Michael, H.A., Neumann, R.B., Beckie, R., Islam, S., Ahmed, M.F., 2006. Groundwater dynamics and arsenic contamination in Bangladesh. Chem. Geol. 228, 112–136. https://doi.org/10.1016/j.chemgeo.2005.11.025

Harvey, R.W., Metge, D.W., LeBlanc, D.R., Underwood, J., Aiken, G.R., Butler, K., McCobb, T.D., Jasperse, J., 2015. Importance of the colmation layer in the transport and removal of cyanobacteria, viruses, and dissolved organic carbon during natural lake-bank filtration. J. Environ. Qual. 44, 1413–1423. https://doi.org/10.2134/jeq2015.03.0151

Hiscock, K.M., Grischek, T., 2002. Attenuation of groundwater pollution by bank filtration. J. Hydrol. 266, 139–144. https://doi.org/10.1016/S0022-1694(02)00158-0

Hoque, M.A., Burgess, W.G., Shamsudduha, M., Ahmed, K.M., 2011. Delineating low-arsenic groundwater environments in the Bengal Aquifer System, Bangladesh. Appl. Geochemistry 26, 614–623. https://doi.org/10.1016/j.apgeochem.2011.01.018

Hoque, M.A., McArthur, J.M., Sikdar, P.K., 2014. Sources of low-arsenic groundwater in the Bengal Basin: investigating the influence of the last glacial maximum palaeosol using a 115-km traverse across Bangladesh. Hydrogeol. J. 22, 1535–1547.

https://doi.org/10.1007/s10040-014-1139-8

Hoque, S.F., Hope, R., Arif, S.T., Akhter, T., Naz, M., Salehin, M., 2019. A social-ecological analysis of drinking water risks in coastal Bangladesh. Sci. Total Environ. 679, 23–34. https://doi.org/10.1016/j.scitotenv.2019.04.359

Hrachowitz, M., Savenije, H.H.G., Blöschl, G., McDonnell, J.J., Sivapalan, M., Pomeroy, J.W., Arheimer, B., Blume, T., Clark, M.P., Ehret, U., Fenicia, F., Freer, J.E., Gelfan, A., Gupta, H. V., Hughes, D.A., Hut, R.W., Montanari, A., Pande, S., Tetzlaff, D., Troch, P.A., Uhlenbrook, S., Wagener, T., Winsemius, H.C., Woods, R.A., Zehe, E., Cudennec, C., 2013. A decade of Predictions in Ungauged Basins (PUB)-a review. Hydrol. Sci. J. 58, 1198–1255. https://doi.org/10.1080/02626667.2013.803183

Hurlimann, A., Dolnicar, S., 2016. Public acceptance and perceptions of alternative water sources: A comparative study in nine locations. Int. J. Water Resour. Dev. 32, 650–673. https://doi.org/10.1080/07900627.2016.1143350

Islam, M.A., Akber, M.A., Ghosh, P.K., 2018. Water quality of small-scale desalination plants in southwest coastal Bangladesh. Water Sci. Technol. Water Supply 18, 1606–1616. https://doi.org/10.2166/ws.2017.222

Islam, M.A., Sakakibara, H., Karim, M.R., Sekine, M., Mahmud, Z.H., 2011. Bacteriological assessment of drinking water supply options in coastal areas of Bangladesh. J. Water Health 9, 415–428. https://doi.org/10.2166/wh.2011.114

Islam, M.M., Chou, F.N.F., Kabir, M.R., Liaw, C.H., 2010. Rainwater: A Potential Alternative Source for Scarce Safe Drinking and Arsenic Contaminated Water in Bangladesh. Water Resour. Manag. 24, 3987–4008. https://doi.org/10.1007/s11269-010-9643-7

Jeong, H.Y., Jun, S.C., Cheon, J.Y., Park, M., 2018. A review on clogging mechanisms and managements in aquifer storage and recovery (ASR) applications. Geosci. J. 22, 667–679. https://doi.org/10.1007/s12303-017-0073-x

Kabir, A., Howard, G., 2007. Sustainability of arsenic mitigation in Bangladesh: Results of a functionality survey. Int. J. Environ. Health Res. 17, 207–218. https://doi.org/10.1080/09603120701254904

Kallali, H., Anane, M., Jellali, S., Tarhouni, J., 2007. GIS-based multi-criteria analysis for potential wastewater aquifer recharge sites. Desalination 215, 111–119. https://doi.org/10.1016/j.desal.2006.11.016

Karim, M.R., Manshoven, S., Islam, M.R., Gascon, J.A., Ibarra, M., Diels, L., Rahman, M.M., 2013. Assessment of an urban contaminated site from tannery industries in Dhaka city, Bangladesh. J. Hazardous, Toxic, Radioact. Waste 17, 52–61. https://doi.org/10.1061/(ASCE)HZ.2153-5515.0000139

Kim, J.W., Choi, H., Pachepsky, Y.A., 2010. Biofilm morphology as related to the porous media clogging. Water Res. 44, 1193–1201. https://doi.org/10.1016/j.watres.2009.05.049

Kjellén, M., McGranahan, G., 2006. Informal Water Vendors And the Urban Poor. International Institute for Environment and Development, London.

Klein, R.J.T., Nicholls, R.J., 1999. Assessment of Coastal Vulnerability to Climate Change. Ambio 28, 182–187.

Knappett, P.S.K., Escamilla, V., Layton, A., McKay, L.D., Emch, M., Williams, D.E., Huq, R., Alam, J., Farhana, L., Mailloux, B.J., Ferguson, A., Sayler, G.S., Ahmed, K.M., van

Geen, A., 2011. Impact of population and latrines on fecal contamination of ponds in rural Bangladesh. Sci. Total Environ. 409, 3174–3182. https://doi.org/10.1016/j.scitotenv.2011.04.043

Kränzlin, I., 2000. Pond management in rural Bangladesh: problems and possibilities in the context of the water supply crisis. Nat. Resour. Forum 24, 211–223. https://doi.org/10.1111/j.1477-8947.2000.tb00945.x

Leung, C.M., Jiao, J.J., 2006. Heavy metal and trace element distributions in groundwater in natural slopes and highly urbanized spaces in Mid-Levels area, Hong Kong. Water Res. 40, 753–767. https://doi.org/10.1016/j.watres.2005.12.016

Levantesi, C., La Mantia, R., Masciopinto, C., Böckelmann, U., Ayuso-Gabella, M.N., Salgot, M., Tandoi, V., Van Houtte, E., Wintgens, T., Grohmann, E., 2010. Quantification of pathogenic microorganisms and microbial indicators in three wastewater reclamation and managed aquifer recharge facilities in Europe. Sci. Total Environ. 408, 4923–4930. https://doi.org/10.1016/j.scitotenv.2010.07.042

Liebe, J.R., Van De Giesen, N., Andreini, M., Walter, M.T., Steenhuis, T.S., 2009. Determining watershed response in data poor environments with remotely sensed small reservoirs as runoff gauges. Water Resour. Res. 45, 1–12. https://doi.org/10.1029/2008WR007369

Lowry, C.S., Anderson, M.P., 2006. An assessment of aquifer storage recovery using ground water flow models. Ground Water 44, 661–667. https://doi.org/10.1111/j.1745-6584.2006.00237.x

MacDonald, A.M., Calow, R.C., 2009. Developing groundwater for secure rural water supplies in Africa. Desalination 248, 546–556. https://doi.org/10.1016/j.desal.2008.05.100

Maeng, S.K., Ameda, E., Sharma, S.K., Grützmacher, G., Amy, G.L., 2010. Organic micropollutant removal from wastewater effluent-impacted drinking water sources during bank filtration and artificial recharge. Water Res. 44, 4003–4014. https://doi.org/10.1016/j.watres.2010.03.035

Maliva, R.G., Guo, W., Missimer, T.M., 2006. Aquifer Storage and Recovery: Recent Hydrogeological Advances and System Performance. Water Environ. Res. 78, 2428–2435. https://doi.org/10.2175/106143006X123102

Martin, R., (ed.) (2013) Clogging issues associated with managed aquifer recharge methods. IAH Commission on Managing Aquifer Recharge, Australia.

Mäusezahl, D., Christen, A., Pacheco, G.D., Tellez, F.A., Iriarte, M., Zapata, M.E., Cevallos, M., Hattendorf, J., Cattaneo, M.D., Arnold, B., Smith, T.A., Colford, J.M., 2009. Solar drinking water disinfection (SODIS) to reduce childhood diarrhoea in rural Bolivia: A cluster-randomized, controlled trial. PLoS Med. 6. https://doi.org/10.1371/journal.pmed.1000125

Mays, D.C., 2013. Clogging in managed aquifer recharge: flow, geochemistry, and clay colloids. In: Martin R (ed.) Clogging issues associated with managed aquifer recharge methods. IAH Commission on Managing Aquifer Recharge, Australia, 14-24

McGranahan, G., Balk, D., Anderson, B., 2007. The rising tide: assessing the risks of climate change and human settlements in low elevation coastal zones. Environ.

Urban. 19, 17–37. https://doi.org/10.1177/0956247807076960

Morse, J.W., Arvidson, R.S., 2002. The dissolution kinetics of major sedimentary carbonate minerals. Earth-Science Rev. 58, 51–84. https://doi.org/10.1016/S0012-8252(01)00083-6

Mukherjee, A., Fryar, A.E., Thomas, W.A., 2009. Geologic, geomorphic and hydrologic framework and evolution of the Bengal basin, India and Bangladesh. J. Asian Earth Sci. 34, 227–244. https://doi.org/10.1016/j.jseaes.2008.05.011

Mulligan, M., Cruz, L.L.S., Pena-Arancibia, J., Pandey, B., Mahé, G., Fisher, M., 2011. Water availability and use across the Challenge Program on Water and Food (CPWF) basins. Water Int. 36, 17–41. https://doi.org/10.1080/02508060.2011.543801

Mulligan, M., 2013. WaterWorld: A self-parameterising, physically based model for application in data-poor but problem-rich environments globally. Hydrol. Res. 44, 748–769. https://doi.org/10.2166/nh.2012.217

Nicholls, R.J., Cazenave, A., 2010. Sea-Level Rise and Its Impact on Coastal Zones. Science (80-.). 328, 1517–1520. https://doi.org/10.1126/science.1185782

Nickson, R., McArthur, J., Burgess, W., Ahmed, K.M., Ravenscroft, P., Rahmanñ, M., 1998. Arsenic poisoning of Bangladesh groundwater. Nature 395, 338–338. https://doi.org/10.1038/26387

Ning, R.Y., 2002. Arsenic removal by reverse osmosis. Desalination 143, 237–241. https://doi.org/10.1016/S0011-9164(02)00262-X

Page, D., Bekele, E., Vanderzalm, J., Sidhu, J., 2018. Managed aquifer recharge (MAR) in sustainable urban water management. Water (Switzerland) 10, 1–16. https://doi.org/10.3390/w10030239

Parkhurst, D.L., Appelo, C.A.J., 1999. User's guide to PHREEQC (Version 2): A computer program for speciation, batch-reaction, one-dimensional transport, and inverse geochemical calculations. Denver.

Paul, B.G., Vogl, C.R., 2011. Impacts of shrimp farming in Bangladesh: Challenges and alternatives. Ocean Coast. Manag. 54, 201–211. https://doi.org/10.1016/j.ocecoaman.2010.12.001

Pedersen, H.D., Postma, D., Jakobsen, R., 2006. Release of arsenic associated with the reduction and transformation of iron oxides. Geochim. Cosmochim. Acta 70, 4116–4129. https://doi.org/10.1016/j.gca.2006.06.1370

Pitt, R., Clark, S., Field, R., 1999. Groundwater contamination potential from stormwater infiltration practices. Urban Water 1, 217–236.

Radloff, K.A., Zheng, Y., Stute, M., Weinman, B., Bostick, B., Mihajlov, I., Bounds, M., Rahman, M.M., Huq, M.R., Ahmed, K.M., Schlosser, P., Geen, A. Van, 2017. Applied Geochemistry Reversible adsorption and fl ushing of arsenic in a shallow , Holocene aquifer of Bangladesh. Appl. Geochemistry 77, 142–157. https://doi.org/10.1016/j.apgeochem.2015.11.003

Rahman, A.K.M.M., Ahmed, K.M., Butler, A.P., Hoque, A.M., 2018. Influence of surface geology and micro-scale land use on the shallow subsurface salinity in deltaic coastal areas: a case from southwest Bangladesh. Environ. Earth Sci. 77, 423. https://doi.org/10.1007/s12665-018-7594-0

Rahman, M.A., Rusteberg, B., Gogu, R.C., Lobo Ferreira, J.P., Sauter, M., 2012. A new spatial multi-criteria decision support tool for site selection for implementation of managed aquifer recharge. J. Environ. Manage. 99, 61–75. https://doi.org/10.1016/j.jenvman.2012.01.003

Rahman, M.M., Hassan, M.Q., Islam, M.S., Shamsad, S.Z.K.M., 2000. Environmental impact assessment on water quality deterioration caused by the decreased Ganges outflow and saline water intrusion in south-western Bangladesh. Environ. Geol. 40, 31–40. https://doi.org/10.1007/s002540000152

Ranjan, P., Kazama, S., Sawamoto, M., 2006. Effects of climate change on coastal fresh groundwater resources. Glob. Environ. Chang. 16, 388–399. https://doi.org/10.1016/j.gloenvcha.2006.03.006

Rao, N.S., Rao, P.S., Dinakar, A., Rao, P.V.N., Marghade, D., 2017. Fluoride occurrence in the groundwater in a coastal region of Andhra Pradesh, India. Appl. Water Sci. 7, 1467–1478. https://doi.org/10.1007/s13201-015-0338-3

Rauch-Williams, T., Hoppe-Jones, C., Drewes, J.E., 2010. The role of organic matter in the removal of emerging trace organic chemicals during managed aquifer recharge. Water Res. 44, 449–460. https://doi.org/10.1016/j.watres.2009.08.027

Ravenscroft, P., Burgess, W.G., Ahmed, K.M., Burren, M., Perrin, J., 2005. Arsenic in groundwater of the Bengal Basin, Bangladesh: Distribution, field relations, and hydrogeological setting. Hydrogeol. J. 13, 727–751. https://doi.org/10.1007/s10040-003-0314-0

Rodell, M., Velicogna, I., Famiglietti, J.S., 2009. Satellite-based estimates of groundwater depletion in India. Nature 460, 999–1002. https://doi.org/10.1038/nature08238

Sarker, M.M.R., Van Camp, M., Islam, M., Ahmed, N., Walraevens, K., 2018. Hydrochemistry in coastal aquifer of southwest Bangladesh: origin of salinity. Environ. Earth Sci. 77, 1–20. https://doi.org/10.1007/s12665-017-7196-2

Sengupta, S., McArthur, J.M., Sarkar, A., Leng, M.J., Ravenscroft, P., Howarth, R.J., Banerjee, D.M., 2008. Do Ponds Cause Arsenic-Pollution of Groundwater in the Bengal Basin? An Answer from West Bengal. Environ. Sci. Technol. 42, 5156–5164. https://doi.org/10.1021/es702988m

Shameem, M.I.M., Momtaz, S., Rauscher, R., 2014. Vulnerability of rural livelihoods to multiple stressors: A case study from the southwest coastal region of Bangladesh. Ocean Coast. Manag. 102, 79–87. https://doi.org/10.1016/j.ocecoaman.2014.09.002

Shamsudduha, M., Taylor, R.G., Ahmed, K.M., Zahid, A., 2011. The impact of intensive groundwater abstraction on recharge to a shallow regional aquifer system : evidence from Bangladesh 901–916. https://doi.org/10.1007/s10040-011-0723-4

Sharma, B., Amarasinghe, U., Xueliang, C., de Condappa, D., Shah, T., Mukherji, A., Bharati, L., Ambili, G., Qureshi, A., Pant, D., Xenarios, S., Singh, R., Smakhtin, V., 2010. The indus and the ganges: River basins under extreme pressure. Water Int. 35, 493–521. https://doi.org/10.1080/02508060.2010.512996

Simmons, G., Hope, V., Lewis, G., Whitmore, J., Gao, W., 2001. Contamination of potable roof-collected rainwater in Auckland, New Zealand. Water Res. 35, 1518–1524. https://doi.org/10.1016/S0043-1354(00)00420-6

Singh, R., Archfield, S.A., Wagener, T., 2014. Identifying dominant controls on hydrologic parameter transfer from gauged to ungauged catchments - A comparative hydrology approach. J. Hydrol. 517, 985–996. https://doi.org/10.1016/j.jhydrol.2014.06.030

Sprenger, C., Hartog, N., Hernández, M., Vilanova, E., Grützmacher, G., 2017. Inventory of managed aquifer recharge sites in Europe : historical development , current situation and perspectives. https://doi.org/10.1007/s10040-017-1554-8

Stefan, C., Ansems, N., 2018. Web-based global inventory of managed aquifer recharge applications. Sustain. Water Resour. Manag. 4, 153–162. https://doi.org/10.1007/s40899-017-0212-6

Sultana, S., Ahmed, K.M., Mahted-Ul-Alam, S.M., Hasan, M., Tuinhof, A., Ghosh, S.K., Rahman, M.S., Ravenscroft, P., Zheng, Y., 2014. Low-Cost Aquifer Storage and Recovery: Implications for Improving Drinking Water Access for Rural Communities in Coastal Bangladesh. J. Hydrol. Eng. 20. https://doi.org/10.1061/(ASCE)HE

Thieme, M., Lehner, B., Abell, R., Hamilton, S.K., Kellndorfer, J., Powell, G., Riveros, J.C., 2007. Freshwater conservation planning in data-poor areas: An example from a remote Amazonian basin (Madre de Dios River, Peru and Bolivia). Biol. Conserv. 135, 484–501. https://doi.org/10.1016/j.biocon.2006.10.054

Umitsu, M., 1993. Late quaternary sedimentary environments and landforms in the Ganges Delta. Sediment. Geol. 83, 177–186. https://doi.org/10.1016/0037-0738(93)90011-S

UNICEF, WHO, 2019. Progress on Drinking Water, Sanitation and Hygiene 2000-2017: Special focus on inequalities. https://doi.org/10.1111 / tmi.12329

United Nations, 2018. Transforming Our World: The 2030 Agenda for Sustainable Development - A/RES/70/1. https://doi.org/10.1891/9780826190123.ap02

van Beek, C.G.E.M., 1984. Restoring Well Yield in the Netherlands. Am. Water Work. Assoc. 76, 66–72.

Van Halem, D., Bakker, S.A., Amy, G.L., Van Dijk, J.C., 2009. Arsenic in drinking water: A worldwide water quality concern for water supply companies. Drink. Water Eng. Sci. 2, 29–34. https://doi.org/10.5194/dwes-2-29-2009

Wada, Y., Van Beek, L.P.H., Van Kempen, C.M., Reckman, J.W.T.M., Vasak, S., Bierkens, M.F.P., 2010. Global depletion of groundwater resources. Geophys. Res. Lett. 37, 1–5. https://doi.org/10.1029/2010GL044571

Wallis, I., Prommer, H., Simmons, C.T., Post, V., Stuyfzand, P.J., 2010. Evaluation of conceptual and numerical models for arsenic mobilization and attenuation during managed aquifer recharge. Environ. Sci. Technol. 44, 5035–5041. https://doi.org/10.1021/es100463q

Ward, J.D., Simmons, C.T., Dillon, P.J., Pavelic, P., 2009. Integrated assessment of lateral flow, density effects and dispersion in aquifer storage and recovery. J. Hydrol. 370, 83–99. https://doi.org/10.1016/j.jhydrol.2009.02.055

Weinman, B., Goodbred, S.L., Zheng, Y., Aziz, Z., Steckler, M., van Geen, A., Singhvi, A.K., Nagar, Y.C., 2008. Contributions of floodplain stratigraphy and evolution to the spatial patterns of groundwater arsenic in Araihazar, Bangladesh. Bull. Geol. Soc. Am. 120, 1567–1580. https://doi.org/10.1130/B26209.1

Werner, A.D., Simmons, C.T., 2009. Impact of sea-level rise on sea water intrusion in coastal aquifers. Ground Water 47, 197–204. https://doi.org/10.1111/j.1745-6584.2008.00535.x

Winkel, L., Berg, M., Amini, M., Hug, S.J., Annette Johnson, C., 2008. Predicting groundwater arsenic contamination in Southeast Asia from surface parameters. Nat. Geosci. 1, 536–542. https://doi.org/10.1038/ngeo254

Worland, S.C., Hornberger, G.M., Goodbred, S.L., 2015. Source, transport, and evolution of saline groundwater in a shallow Holocene aquifer on the tidal deltaplain of southwest Bangladesh. Water Resour. Res. 51, 5791–5805. https://doi.org/10.1002/2014WR016262

World Health Organization, 2006. Guidelines for drinking-water quality [electronic resource] : incorporating first addendum. Vol. 1, Recommendations. – 3rd ed. Geneva. https://doi.org/10.1016/S1462-0758(00)00006-6

World Health Organization, UNICEF, 2017. Progress on Drinking Water, Sanitation and Hygiene - 2017 update and SDG baselines. Geneva. https://doi.org/10.1111 / tmi.12329

Yokota, H., Tanabe, K., Sezaki, M., Akiyoshi, Y., Miyata, T., Kawahara, K., Tsushima, S., Hironaka, H., Takafuji, H., Rahman, M., Ahmad, S.A., Sayed, M.H.S.U., Faruquee, M.H., 2001. Arsenic contamination of ground and pond water and water purification system using pond water in Bangladesh. Eng. Geol. 60, 323–331. https://doi.org/10.1016/S0013-7952(00)00112-5

Zaman, A.M., Molla, M.K., Pervin, I.A., Rahman, S.M.M., Haider, A.S., Ludwig, F., Franssen, W., 2017. Impacts on river systems under 2 °C warming: Bangladesh Case Study. Clim. Serv. 7, 96–114. https://doi.org/10.1016/j.cliser.2016.10.002

Zuurbier, K.G., Bakker, M., Zaadnoordijk, W.J., Stuyfzand, P.J., 2013. Identification of potential sites for aquifer storage and recovery (ASR) in coastal areas using ASR performance estimation methods. Hydrogeol. J. 21, 1373–1383. https://doi.org/10.1007/s10040-013-1003-2

CHAPTER 2

GROUNDWATER SALINITY VARIATION IN UPAZILA ASSASUNI (SOUTHWESTERN BANGLADESH), AS STEERED BY SURFACE CLAY LAYER THICKNESS, RELATIVE ELEVATION AND PRESENT-DAY LAND USE

Floris Loys Naus

Paul Schot

Koos Groen

Kazi Matin Ahmed

Jasper Griffioen

2019. Hydrological and Earth System Sciences 23, 1431–1451.

ABSTRACT.

In the southwestern coastal region of Bangladesh, options for drinking water are limited by groundwater salinity. To protect and improve the drinking water supply, the large variation in groundwater salinity needs to be better understood. This study identifies the palaeo and present-day hydrological processes and their geographical or geological controls that determine variation in groundwater salinity in Upazila Assasuni in southwestern Bangladesh. Our approach involved three steps: a geological reconstruction, based on the literature; fieldwork to collect high-density hydrological and lithological data; and data processing to link the collected data to the geological reconstruction in order to infer the evolution of the groundwater salinity in the study area. Groundwater freshening and salinization patterns were deduced using PHREEQC cation exchange simulations and isotope data were used to derive relevant hydrological processes and water sources. We found that the factor steering the relative importance of palaeo and present-day hydrogeological conditions was the thickness of the Holocene surface clay layer. The groundwater in aquifers under thick surface clay layers is controlled by the palaeohydrological conditions prevailing when the aquifers were buried. The groundwater in aquifers under thin surface clay layers is affected by present-day processes, which vary depending on present-day surface elevation. Slightly higher-lying areas are recharged by rain and rainfed ponds and therefore have fresh groundwater at shallow depth. In contrast, the lower-lying areas with a thin surface clay layer have brackish–saline groundwater at shallow depth because of flooding by marine-influenced water, subsequent infiltration and salinization. Recently, aquaculture ponds in areas with a thin surface clay layer have increased the salinity in the underlying shallow aquifers. We hypothesize that to understand and predict shallow groundwater salinity variation in southwestern Bangladesh, the relative elevation and land use can be used as a first estimate in areas with a thin surface clay layer, while knowledge of palaeohydrogeological conditions is needed in areas with a thick surface clay layer.

2.1 INTRODUCTION

In the Ganges–Brahmaputra–Meghna (GBM) river delta, home to 170 million people, availability of safe drinking water is problematic because of the very seasonal rainfall, the likelihood of arsenic occurrence in the shallow groundwater and the pollution of surface water bodies (Harvey et al., 2002; Ravenscroft et al., 2005; Chowdhury, 2010; Sharma et al., 2010; Bhuiyan et al., 2011). In the southwestern coastal region of Bangladesh, suitable drinking water options are even more limited, as here the groundwater is largely brackish to saline (Bahar and Reza, 2010; George, 2013; Fakhruddin and Rahman, 2014; Worland et al., 2015; Ayers et al., 2016). Consequently, the people in this region are at high risk of preeclampsia, eclampsia and gestational hypertension from drinking groundwater, and at increased risk of ingesting pathogens from water from the traditional ponds (Kränzlin, 2000; Khan et al., 2014). Stress on the limited reserves of fresh groundwater is expected to rise in the future through a combination of climate change, sea level rise, increased abstraction for irrigation and industry, and population growth (Shameem et al., 2014; Auerbach et al., 2015). To protect and improve the drinking water supply in the coastal region it is therefore important to understanding the present-day spatial variation and formation processes of the groundwater salinity.

Previous studies have found great variation in the groundwater salinity in southwestern Bangladesh (BGS and DHPE, 2001; George, 2013; Worland et al., 2015; Ayers et al., 2016). Several explanations for this large variation have been proposed. One is present-day saline water recharge from the tidal rivers and creeks and the aquaculture ponds that cover much of the region (Rahman et al., 2000; Bahar et al., 2010; Paul and Vogl, 2011; Ayers et al., 2016). Another is freshwater recharge where the clay cover is relatively thin and from rainfed inland water bodies (George, 2013; Worland et al., 2015; Ayers et al., 2016). Finally, some of the variation in the salinity of the groundwater is thought to reflect historical conditions prevailing when the aquifer was buried (George, 2013; Worland et al., 2015; Ayers et al., 2016). Studies in coastal deltas elsewhere, where a higher head due to freshwater infiltration in the higher areas leads to the formation of freshwater lenses, have identified elevation differences as being important factors controlling groundwater salinity (Stuyfzand, 1993; Walraevens et al., 2007; Goes et al., 2009; de Louw et al., 2011; Santos et al., 2012). It has been suggested that both present-day and palaeohydrological processes are important, as deltas are almost never in equilibrium with present-day boundary conditions (Sukhija et al., 1996; Groen et al., 2000; Post and Kooi, 2003; Sivan et al., 2005; Delsman et al., 2014).

It remains unclear how each of the proposed processes influences groundwater salinity variation in southwestern Bangladesh. Previous studies found no spatial autocorrelation in groundwater salinity, presumably because the sampling distances were larger than the expected variation in groundwater salinity (Ayers et al., 2016). In our study, we set out to elucidate the hydrological processes that determine the salinity variation in the groundwater by using high-density sampling in a case study area with large variation in land use, surface water bodies and surface elevation. In addition, we aimed to identify geographical or geological factors controlling the dominant salinization and freshening processes and, therefore, the groundwater salinity.

2.2 METHODS

2.2.1 Methodological approach

Our approach consisted of three steps. First, based on the literature, we reconstructed the geological evolution of southwestern Bangladesh. Second, in the field we carried out high-density hydrological and lithological data collection to capture the large expected variation in groundwater salinity. Third, we inferred the evolution of the groundwater salinity by interpreting the field data in the light of the regional geological reconstruction and present-day surface conditions, to determine the dominant processes responsible for the variation in groundwater salinity.

2.2.2 Fieldwork

To research all the proposed processes, the study area needed to have much variation in land use and surface water bodies, and appreciable variation in surface elevation.We selected an area in southwestern Bangladesh by analysing satellite imagery (World Imagery, ESRI, Redlands, CA, USA) to distinguish land use and surface water bodies, Shuttle Radar Topography Mission data (SRTM) (Farr et al., 2007) to analyse elevation patterns and a soil map (FAO, 1959) to ascertain surface geology and geomorphology. The case study area, which is in the Assasuni Upazila (Figure 2.1), comprises settlements on slightly higher land, surrounded by lower-lying agricultural fields and aquaculture ponds (Figure 2.1). There are freshwater ponds in the settlements. The soil in the study area is composed of fluvial silts in the higher areas and tidal flat clays in the surrounding lower-lying areas (FAO, 1959). The study was conducted along a crooked 6.3 km long transect oriented approximately north–south (Figure 2.1) running through several settlements.

At the northern and southern ends are tidal creeks (Figure 2.1) whose salinity varies seasonally. They are fresh in the monsoon period, but the salinity slowly rises during the dry season and by April and May the creek water contains up to two-thirds seawater (Bhuiyan et al., 2012).

In 2017, hydrological and lithological data were collected along the transect at high density (see Sect. 2.2.1) to a depth of 50 m in two field campaigns: one in the dry season (January–February) and one in the wet season (July–August). To do so, groundwater observation wells were constructed, the ground was levelled, and surface and groundwater were sampled. The hydrological data were used to establish (1) the present-day variation in groundwater salinity (2) whether groundwater is affected by freshening or salinization (this entailed analysing the cation exchange) and (3) the source water type recharging the groundwater (for this we analysed the isotopic data). The lithological data were linked with the reconstructed geological history to determine the palaeohydrogeological conditions in the study area from the Last Glacial Maximum (LGM) until the present day.

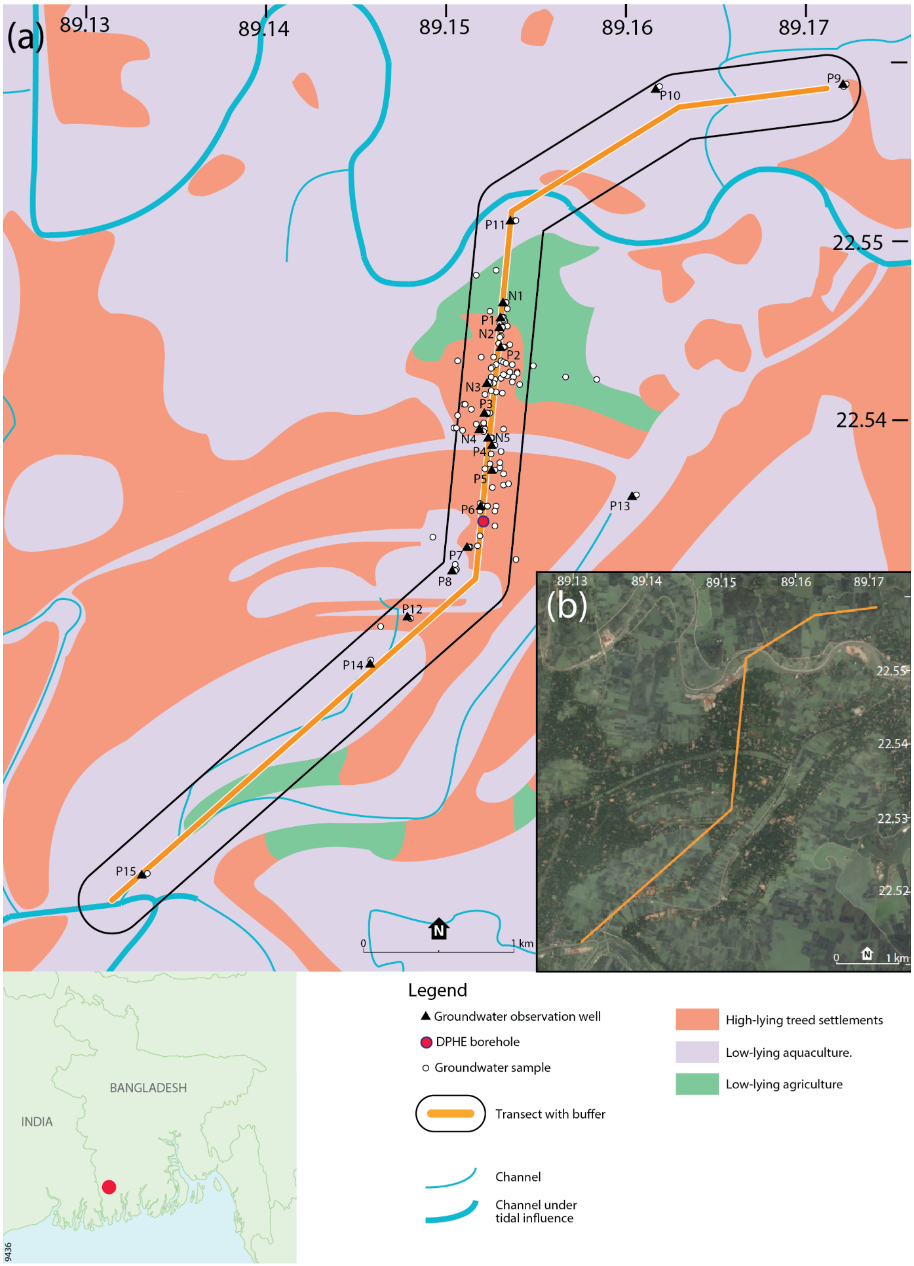


Figure 2.1. (a) Overview of the study area, indicating the transect and groundwater observation points, land use and current surface water channels. The land use types are based on satellite imagery (b), SRTM data, a soil map (FAO, 1959) and field observations. The location of the Department of Public Health Engineering (DPHE) borehole is indicated in the figure. (b) Satellite imagery of the study area (ESRI, 2018).

2.2.2.1 Lithological drillings and groundwater observation wells

Groundwater observation wells were installed to collect lithological information and groundwater samples. In 2017, 34 tubes with filters at depths of between 6 and 46 m deep were installed at 20 locations. At 14 locations, a single tube was installed, at four other locations, a nest of two tubes was installed, and at the last four locations, a nest of three tubes was installed. Two groundwater observation wells (P16 and P17) were installed a year later to get more detailed lithological information between P11 and P10, but as the sampling campaign had ended, no water samples are available for them.

The first drilling at each location was used to collect the lithological data to approximately 46 m depth (150 feet). The traditional “sludger” or “hand-flapper” method was used as drilling technique (Horneman et al., 2004). The drilling fluid was water from nearby surface water or tube wells, which was pumped out directly after installation by pumping the tube wells for at least 30 min or until EC (electrical conductivity) and temperature had stabilized. During the first drilling at each location, the sediment slurry was interpreted in the field every 1.524 m (5 feet). Additional lab analyses were performed on 47 sediment samples from the surface clay layer and at the filter depths. These samples were analysed for their grain size distribution with a Malvern Scirocco 2000, after pre-treatment to remove organic matter and carbonates and after peptizing the mud particles using a peptization fluid and ultrasound. The particles less than 8 μm were classed as clay, those between 8 and 63 μm as silt and those over 63 μm as sand (Konert and Vandenberghe, 1997).

The carbonate and organic matter contents of the sampleswere quantified by thermogravimetric analysis (TGA) using a Leco TGA-601. The percentage of organic matter was defined as the weight loss percentage between 150 and 550 °C, corrected for structural water loss from clay by a factor of 0.07 times the fraction smaller than 8 μm (van Gaans et al., 2010; Hoogsteen et al., 2015). The carbonate content was determined as the percentage weight loss between 550 and 850 °C.

Table 2.1. Overview of chemical and isotope samplings.

	Samples for IC and ICP-MS (N = 129)	Samples for δ^2H and $\delta^{18}O$ (N = 45)	Samples for tritium (N = 23)
Dry season sampling campaign (January and February 2017)	26 groundwater observation wells, 68 household tube wells, 10 freshwater ponds, 2 aquaculture ponds, 8 open auger borings	-	-
Wet season sampling campaign (July and August 2017)	6 groundwater observation wells, 2 freshwater ponds, 3 aquaculture ponds, 3 open auger borings, 1 inundated field	27 groundwater observation wells, 9 household tube wells, 2 freshwater ponds, 3 aquaculture ponds, 3 open auger boring, 1 inundated field	14 groundwater observation wells, 6 household tube wells, 3 open auger borings

2.2.2.2 Elevation

Using a Topcon ES series total station (Topcon, Japan), surface elevation at the installed groundwater observation wells was measured relative to a zero benchmark (a concrete slab at nest 1). The elevation data were used to correlate the wells in terms of their water levels as measured at least 2 days after installation.

2.2.2.3 Hydrochemistry and isotopes

During two sampling campaigns, water samples were taken and analysed for anions (IC) and cations (ICP-MS); samples were also taken for tritium analysis, as well as for δ^2H and $\delta^{18}O$ analyses. For details, see Table 2.1. The groundwater in the groundwater observation wells was sampled at least a week after installation. The household tube wells and groundwater observation wells were purged by pumping approximately three times the volume inside the tube. To sample porewater from the clay, we used the open auger boring method (De Goffau et al., 2012). We drilled a hole with a hand auger, without any drilling fluid. Then we waited for the hole to fill with water, which we sampled by inserting a sample bottle into the hole with a stick, which we pulled back out using a rope that was attached to the bottle. EC, temperature and pH were measured directly in the field using a HANNA HI 9829 (Hanna Instruments, USA). Alkalinity was determined by titration within 36 hours of sampling (Hach Company, USA). The samples were kept out of the sun and cooled as much as possible.

For the IC and ICP-MS analyses, the water samples were stored in a 15 ml polyethylene tube after filtering through a 0.45 μm membrane. Back in the Netherlands, aliquots were transferred to 1.5 ml glass vials with septum caps for IC analysis. For the IC, the aliquots were diluted in accordance with their EC, which was used as an approximation of their salinity. Below 2000 μS/cm the aliquots were not diluted (1:0), between 2000 and 4000 μS/cm the aliquots were diluted two times (1:1), between 4000 and 10000 μS/cm the aliquots were diluted five times (1:4), and above 10000 μS/cm the aliquots were diluted ten times (1:9). The remaining sample was spiked by adding 100 μl of nitric acid (HNO_3), put on a shaker for 72 hours, and used for ICP-MS. For the ICP-MS, the chloride concentrations were low enough to allow for direct measurement. However, samples 34, 128, 149, 150, 151, 164, 167, 169, 176, 178 showed matrix effects, so were remeasured after diluting five times. The samples for isotope analysis (δ^2H and $\delta^{18}O$) were stored in 15 ml polyethylene tubes and analysed on a Thermo GasBench-II coupled to a Delta-V advantage (Thermo Fisher Scientific, USA). Samples for tritium analysis were stored in polyethylene 1 litre bottles and analysed according to NEN-EN-ISO 9698.

2.2.3 Calculations and modelling

2.2.3.1 Variations in groundwater salinity

To study salinity variation the water samples were classified on the basis of chloride concentration into four classes (adjusted from Stuyfzand, 1993): fresh (chloride concentration <150 mg/L) brackish (chloride concentration 150–1000 mg/L), brackish–saline (chloride concentration 1000–2500 mg/L) and saline (chloride concentration >2500 mg/L). We estimated the chloride values of observation wells P16 and P17 from their EC values: we assigned them the chloride values of samples with similar EC values.

2.2.3.2 Interpretation of isotopic composition

The stable isotopic composition was interpreted using the mixing line between rainwater and seawater and the meteoric water lines of the two closest meteorological stations with isotopic data: Dhaka and Barisal (respectively 185 and 125 km from the study area) (IAEA/WMO, 2017). For the mixing line between rain and seawater, the weighted average rainwater composition was based on data from the Barisal meteorological station (IAEA/WMO, 2017) and the seawater isotopic composition was based on Vienna Standard Mean Ocean Water (VSMOW).

2.2.3.3 Cation exchange

Evidence of cation exchange was assessed by calculating the amount of enrichment or depletion of the cations compared to conservative mixing for each sample. Chloride was used as an indicator of the degree of conservative mixing. The deviation from conservative mixing for compound i (in meq/L) was calculated using the following formulae, based on Griffioen (2003):

$$iZ = i_{sample} - i_{conservative} \tag{1}$$

with:

$$i_{conservative} = i_{fresh} + \left(i_{sea} - i_{fresh}\right) \cdot \frac{\left(Cl_{sample} - Cl_{fresh}\right)}{\left(Cl_{Sea} - Cl_{fresh}\right)} \tag{2}$$

where *i* refers to the concentration in meq/L. Seawater is used for the saline end member i_{sea}. To calculate the Z-values we used values from Ganges water (Sarin et al., 1989) and from pond water (this study) for the freshwater end member i_{fresh}. We assumed that Ganges water has had a large influence on the study area for most of the Holocene and that pond water might have influenced the groundwater recently. Z-values had to be negative or positive for both freshwater end members to be accepted as being truly affected by hydrogeochemical processes. To account for false positive or false negative Z-value due to errors in analysis, the Z-values also had to be larger than the expected error for them to be interpreted as affected by hydrogeochemical processes. Like Griffioen (2003), we assumed the expected error in the amount of exchange was 2.8%, based on a standard error in analysis of 2% and standard propagation of error. The same formula was used to indicate whether sulphate had been depleted by reduction or enriched by other sources.

2.2.3.4 PHREEQC simulations of cation exchange

For the interpretation of the hydrogeochemical processes that have occurred in each of the groundwater samples, we needed to take account of site-specific conditions and site-specific hydrochemical processes; to do so, we used the PHREEQC model code (Parkhurst and Appelo, 2013). Possible dissolution or precipitation of minerals was assessed by calculating saturation indices for calcite, dolomite and gypsum, and the partial pressure for CO2. Additionally, cation exchange during salinization or freshening was simulated, to interpret the stage of salinization or freshening for the samples. For salinization, a scenario was simulated in which seawater diluted 10 times displaces Ganges water (Sarin et al., 1989). For freshening, two scenarios were simulated, because different cation exchange patterns were expected for a) a scenario in which Ganges water displaces 10 times diluted sea water and b) a scenario in which Ganges water displaces 100 times diluted sea water. The salinities assigned to the saline water end members were based on the salinity levels of mostly less than 1/10th seawater detected in the groundwater.

The Z-values of the samples were compared to the Z-values calculated in the PHREEQC scenarios. Freshening or salinization was determined based on the NaZ value of the samples, with a positive NaZ value indicating freshening and a negative NaZ value indicating salinization. Next, the simulated MgZ patterns in the three PHREEQC scenarios were used to differentiate between the stages of freshening or salinization in the groundwater samples.

The cation exchange processes were simulated using 1D reactive, advective/dispersive transport. The time steps were 1 year and the groundwater velocity was exactly one cell per time step, which makes the Courant number 1. The dispersivity was taken as half of the velocity, which resulted in a Peclet number of 2. The Courant and Peclet numbers were both within the boundaries of a stable model (Steefel & MacQuarrie 1996). The Cation Exchange Capacity (CEC) used in the model was based on the value calculated from the empirical variables for marine soils given by Van der Molen (1958):

$$CEC = 6.8 * A + 20.4 * B \tag{3}$$

where *A* is the percentage of the particles smaller than 8 μm and *B* is the organic matter percentage, based on values for the 26 sand samples taken in our study. Sulfate reduction and methane production were simulated by introducing CH_2O in a zero-order reaction. For the salinizing scenario, CH_2O was introduced at 0.1 mmol per year. For the freshening scenario, 0.05 mmol per year were introduced. Based on the calculated saturation indices, calcite dissolution was simulated by keeping the calcite saturation index at 0.25 throughout the run, which is representative for marine water (Griffioen 2017).

2.3 RESULTS

2.3.1 Regional hydrogeological reconstruction

The relevant Holocene sedimentary history of the researched upper 50 m of the subsurface starts in a landscape determined by conditions under the LGM. During the LGM, the sea level was much lower than it is today, leading to fresh conditions. The freshwater rivers eroded deeply incised valleys down to 120 m below the present-day land surface (BGS & DPHE, 2001; Hoque et al., 2014; Mukherjee et al., 2009). In the interfluvial areas, a Pleistocene palaeosol formed, characterized by oxidized sands (Umitsu, 1993; Burgess et al., 2010; Hoque et al., 2014). The LGM conditions in the study area are uncertain, as the area is near the edge of a possible palaeo-channel (Hoque et al., 2014; Goodbred et al., 2014), making it possible that the starting conditions for the Holocene sedimentation could be either a Pleistocene incised valley or a palaeosol.

The Holocene sedimentary history in southwestern Bangladesh can be divided into three distinct periods. First, at around 10–11 kyr BP (kilo years before present), a transgressive period started, when the sea level started to rise rapidly (Islam and Tooley, 1999). In combination with an increase in monsoon intensity (Goodbred and Kuehl, 2000b), this marks the start of a period with very rapid sedimentation (Goodbred and Kuehl, 2000a), leading to a transgressive sediment thickness of 20 to 50 m in nearby study sites (Sarkar et al., 2009; Ayers et al., 2016). The maximum inland location of the shoreline was either slightly south or slightly the north of our study area (Goodbred and Kuehl, 2000a; Shamsudduha and Uddin, 2007). During the transgression, sedimentary conditions are expected to have become more under the influence of marine salinity. In the second period – from 8 kyr BP – sea level rise slowed down, and at ~7 kyr BP the coast started to prograde (Goodbred and Kuehl, 2000a; Sarkar et al., 2009; Goodbred et al., 2014), which probably reduced the influence of marine water. Concomitantly, the monsoon intensity decreased, which caused the sedimentation rate to decline (Goodbred and Kuehl, 2000b; Sarkar et al., 2009). Finally, between 5 kyr and 2.5 kyr BP the Ganges moved eastwards (Allison et al., 2003; Goodbred and Kuehl, 2000a; Goodbred et al., 2003; Goodbred et al., 2014; Morgan and McIntire, 1959; Sarkar et al., 2009). The probable cause of the migration was a topographical gradient resulting from disproportional sedimentation by the Ganges in the west part of the delta (Goodbred et al., 2014), which reduced the supply of sand to the study area, resulting in smaller channels and a larger area of floodplain. In these tidal floodplains, silts and clays were deposited during high water events (Allison and Kepple, 2001), which is why clay overlies all of southwest Bangladesh (BGS and DPHE, 2001; Sarkar et al., 2009; Ayers et al., 2016). Some small late Holocene channels depositing fine sand were still present; sediments from such channels have been found in nearby study areas (Sarkar et al., 2009; Ayers et al., 2016).

Even though the general trend since approximately 7 kyr BP has been progradation, there was a period between 4.5 kyr and 2 kyr BP in which the sea level was higher than it is today (Gupta and Amin., 1974; Mathur et al., 2004; Sarkar et al., 2009), which might have led to marine deposition at an elevation above present-day sea level (Goodbred et al., 2003; Sarkar et al., 2009).

2.3.2 Lithology

The collected lithological data reveals a large variation in the thickness and organic matter content of the surface clay layer. In the floodplains in the north at P9, P10, P16 and P17, and in the south at P15, the clay cover is approximately 35 m thick and rich in organic matter (Figure 2.2, Table A2.1), whereas around the settlement in the centre of the transect (henceforth referred to as the central settlement) it is 3–10 m thick and less rich in organic matter (Figure 2.2, Table A2.1). From the middle of the transect towards the north and south, the clay cover becomes gradually thicker (Figure 2.2). Under the clay cover is an aquifer composed of grey sands with carbonates, which extends down to the end of all the drillings at 46 m depth (Figure 2.2, Table A2.1). This main aquifer contains some small discontinuous organic-matter-rich clay layers (Figure 2.2, Table A2.1). Extrapolating from a log of the Department of Public Health Engineering (DPHE) for a borehole located between P6 and P7 (Figure 2.2), it seems likely that this sand layer extends to 110 m depth and is followed by a clay layer from 110 to 128 m depth and a second aquifer down to a depth of 152 m. At N5, the surface clay layer was succeeded by a gravel bed at 10 m depth.

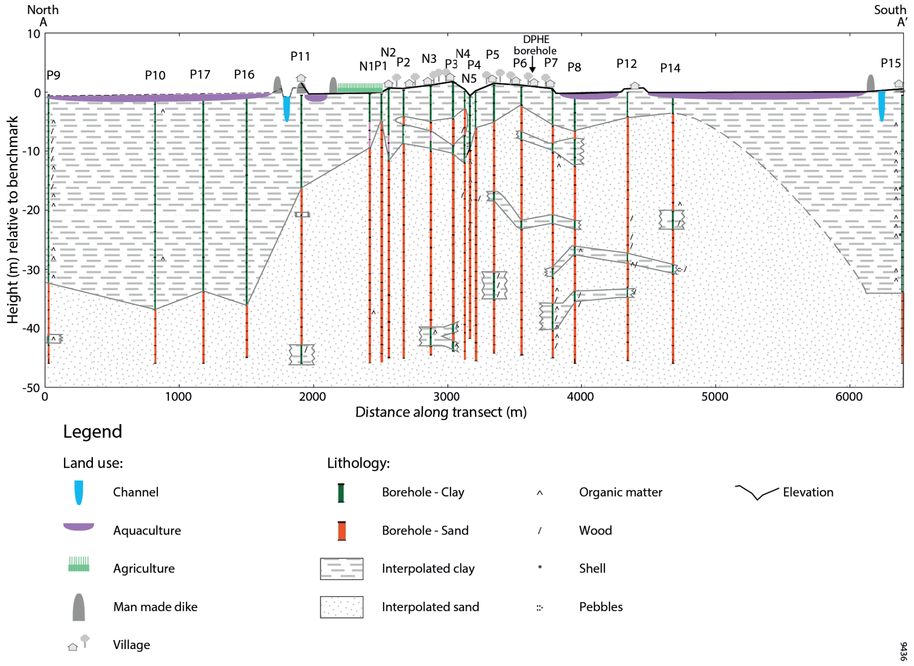


Figure 2.2. Lithological data from the boreholes drilled in this study, and the interpolated sand and clay layers. A clear difference in thickness of the clay layer is visible between the palaeo floodplains at the north and south sides of the transect, and the palaeo channel in the middle of the transect.

2.3.3 Elevation

The villages are at a different elevation than the rest of the study area. Compared with the benchmark, the elevation of the groundwater observation wells in the villages (N2, P2, N3, P3, P5, P6, P7, P11, P12, P15) is between 0.5 to 1.8 m higher, while the elevation in the agricultural fields (N1, P1, N5) and aquaculture ponds (P8, P14) is -0.6 to 0 m (Figure 2.2). The elevation was not measured at P9, P10, P16 or P17 but data from the SRTM and field observations suggest that these areas are also relatively low.

2.3.4 Salinity

The variation in surface water and groundwater salinity is shown in Figure 2.3. Surface and groundwater salinity are discussed separately below.

2.3.4.1 Surface water salinity

The sampled surface water ponds can be divided into fresh and saline ponds. The ponds used for aquaculture contain slightly less than 25% seawater and have a chloride concentration of around 4000 mg/L, and they are saline throughout the year. The rainwater ponds in the settlements on higher land have a chloride concentration below 50 mg/L and are fresh throughout the year. In the wet season, additional surface water bodies are formed, when many of the agricultural fields become flooded by the large amount of rain. The water in the flooded agricultural field near N1 and P1 had a chloride concentration around 200 mg/L in July 2017 (Figure 2.3), which may be caused by the dissolution of salts from the saline topsoil, as salt deposits are visible on the surface after the fields dry out again in the post-monsoon period.

2.3.4.2 Groundwater salinity

The chloride concentrations of the groundwater samples vary between 18 mg/L and 4545 mg/L, which indicates that the most saline groundwater samples contain somewhat less than 25% seawater. The salinity of the groundwater in the first aquifer correlates well with surface elevation: higher areas are fresher than lower areas (Figure 2.3). At the slightly higher central settlements, the groundwater is fresh to a depth of approximately 30 m. Below that depth, the groundwater is brackish or brackish–saline. In the lower areas with a thick clayey top layer at the northern and southern ends of the transect, the groundwater is brackish to brackish–saline (Figure 2.3). This was also the case at P16 and P17, where the EC values were respectively 2.49 and 3.4 mS/cm, which – based on the chloride concentration of samples with similar EC values – corresponds with chloride values between 500 and 1000 mg/L.

In low areas with a thin clay cover, groundwater is saline–brackish under the agricultural areas and saline under the aquaculture ponds. The groundwater salinity difference between these two land use types is substantial: groundwater chloride concentration under the aquaculture ponds exceeds 4000 mg/L, which is more than double the groundwater chloride concentration under the agricultural areas (1000–2000 mg/L). The few samples taken to the side of the transect show the same differences in groundwater salinity between these land use types. The salinity of water samples taken a few metres below the surface from the clay is generally similar to that of the groundwater in the sand aquifer immediately below, except for two clay water samples near the former creek in the middle of the central settlement and one clay water sample from the agricultural fields north of the central settlement (Figure 2.3).

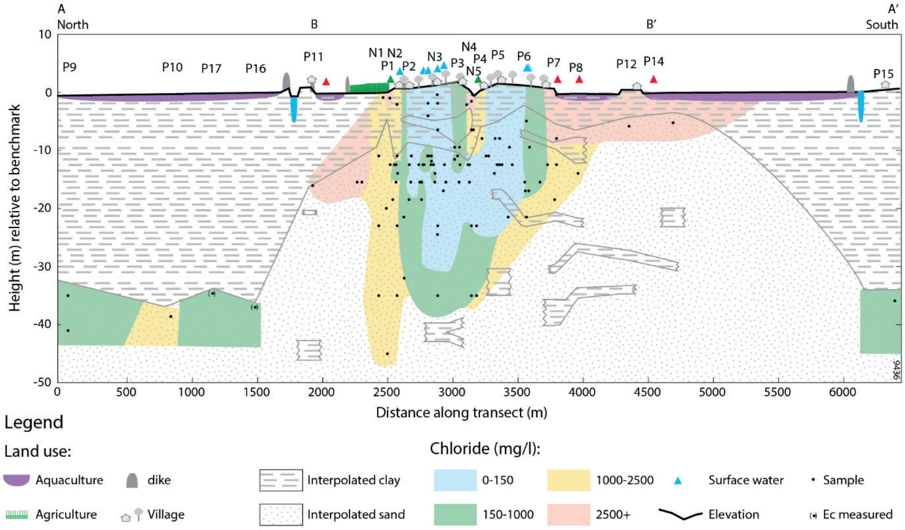


Figure 2.3. Salinity of the sampled groundwater, of phreatic water sampled in clay just below the surface and of sampled surface water bodies.

2.3.5 Stable isotopes

The samples were divided into four source water classes (Figure 2.4). The first class consisted of samples with a relatively light isotopic composition, similar to the weighted average rainwater. This indicates that direct rain infiltration is the dominant source of this water. These samples were taken from surface water bodies in the wet season. The shallow groundwater sample from the clay layer near N3 also falls in this class, revealing that direct infiltration of rain occurs to some extent in the higher-lying area. Lastly, the groundwater at P7 and P8, and at P9, P10 and P15 falls in this class, but since the samples were taken at great depths, or the isotopic composition of overlying groundwater is different, it is unlikely this groundwater formed under present-day conditions (Figure 2.5).

The second class contains samples with a relatively heavy isotopic composition skewed to the right of the MWLs (Figure 2.4). This indicates an effect of evaporation and mixing with seawater. Samples of this class were taken at relatively shallow depths and close to surface water bodies (Figure 2.5), which suggests that the main source of this water is water infiltrating from stagnant surface water.

The third class comprises samples with a relatively heavy isotopic composition located close to the MWLs in Figure 2.4. In the study area, this class is visible in five groundwater samples taken just north of the central settlements (Figure 2.5).

The main water source of this class is unclear, but seems to be relatively heavy rain, with limited evaporation or mixing with seawater.

The last and largest group is made up of intermediate weight samples without one clear water source type. These samples could be a mix of different sources of water, such as rainwater, water from surface water bodies and seawater. Some influence of surface water bodies is indicated by the small skew to the right from the MWLs. Samples from this class were collected from both fresh and saline water under the thin clay layer (Figure 2.5).

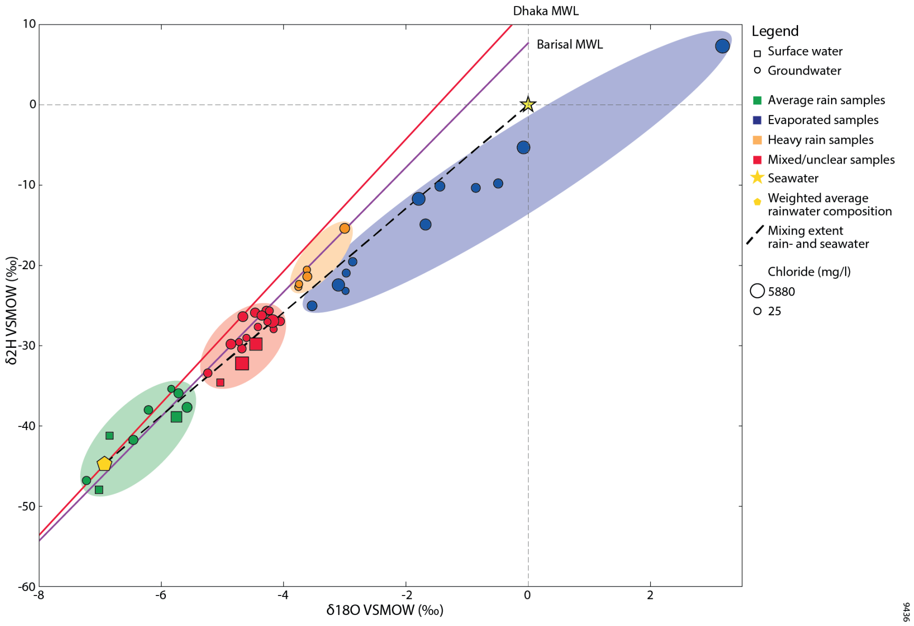


Figure 2.4. Stable isotope content in the study area. Local Meteoric Water Lines based on monthly samples from the two closest stations Dhaka and Barisal at respectively 185 km and 125 km from the study area (IAEA, 2017). The mixing line between rainwater and seawater is based on the weighted average rainwater composition from Barisal (IAEA, 2017) and seawater VSMOW.

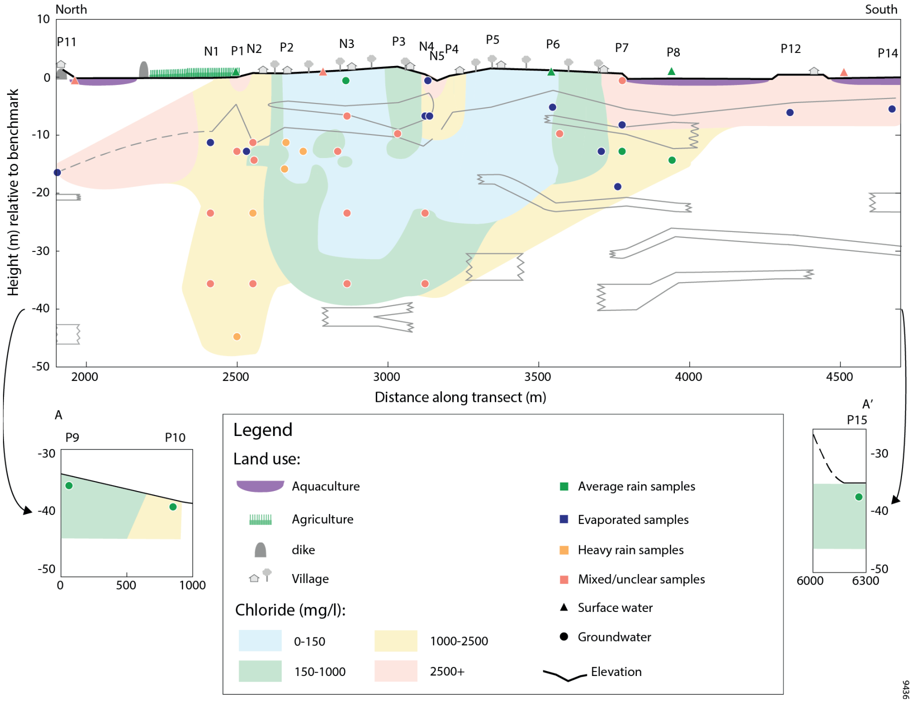


Figure 2.5. Isotope group of the groundwater sampled in the cross section.

2.3.6 Tritium

All 23 samples had a tritium concentration below the detection threshold of 1.64 Bq/L, and therefore tritium content could not be used to date the water. Seawater (<0.4 Bq/L) (Kakiuchi et al., 1999), recent rainwater (0.5 Bq/L), or water older than 70 years would all have a tritium concentration below detection threshold, and might therefore be the source of the groundwater (IAEA/WMO, 2017).

2.3.7 Redox conditions and saturation indices

The conditions in the groundwater are reduced, with sulfate reduction and organic matter decay. Sulfate reduction is indicated by depleted sulfate concentrations compared to conservative mixing. Enrichment of SO_4 is observed only in the shallow saline clay near N4 and N5, possibly due to pyrite oxidation. Organic matter decay can be inferred from the partial pressure of CO_2 varying between $10^{-0.4}$ and $10^{-1.6}$ atm in most of the groundwater samples, which is high even for tropical conditions, but not unusual in Holocene coastal regions (Appelo and Postma, 2005; Griffioen et al., 2013).

Most of the groundwater samples from between 10 and 25 m deep are somewhat supersaturated for calcite, with saturation indices between 0 and 0.7, which is common for seawater-derived groundwater in coastal aquifers (Rezaei et al., 2005; Griffioen et al., 2013). Since the sediments contain carbonates (Table A2.1), calcite, aragonite or dolomite is available for dissolution. The samples that were subsaturated for calcite were taken from the clay cover and the shallow aquifer (<10 m deep).

2.3.8 PHREEQC simulations

The Z-values of the groundwater samples were compared with the patterns of the Z-values during the salinization and freshening scenarios (Figure 2.3). The patterns of the Z-values were used for the interpretation, as the exact Z-values of the samples were expected to be lower than the Z-values in the model scenarios, since each sample is the result of a specific, less extreme mix of end members. The CaZ, MgZ and NaZ of the groundwater samples were plotted as points in the scenario whose Z-value patterns they best matched, with the X location determined by the chloride concentration matching the values of the chloride in the model scenario. Samples with a chloride concentration exceeding the chloride values in the scenario were plotted at the saline sides of the figures. Six cation exchange (CE) groups were identified (Table 2.2).

Table 2.2. Cation exchange groups identified.

Symbol CE group	NaZ value	MgZ value	Description
+	Positive	Positive	Freshening fresh (<200 mg Cl/L)
*	Positive	Negative	Freshening saline (<2000 mg Cl/L)
.	Neutral	Negative, neutral or positive	No cation exchange
-	Negative	Positive	Initial salinization
~	Negative	Neutral	Intermediate salinization
—	Negative	Negative	Late-stage salinization

2.3.8.1 Freshening

The two freshening scenarios show different patterns for MgZ, because the saline water had a larger percentage of Mg on the cation exchange complex than the fresh water (Appelo and Willemsen, 1987; Beekman and Appelo, 1991; Griffioen, 2003). Consequently, the MgZ remained positive during the freshening in the freshwater freshening scenario, while the MgZ became negative during the saline water freshening scenario. We therefore used the MgZ values of the samples to differentiate between freshening freshwater samples (MgZ+) and freshening saline water samples (MgZ-).

Freshening freshwater samples came from multiple locations in the shallow (<20 m deep) fresh groundwater in the central settlement, which indicates that this fresh groundwater is likely formed by water infiltrating from the surface. The freshening detected in the deep brackish samples at P9 and P15 is unlikely to be caused by infiltrating fresh surface water, because of the thick impervious clay layer underlying the saline aquaculture water (Figure 2.7). Like their isotopic composition, this suggests that this groundwater formed under palaeohydrological conditions.

Freshening saline water was detected in three saline samples taken under the agricultural field north of the central village (Figure 2.7). The freshening results either from infiltrating surface water, or from fresh water flowing north, as suggested by head data measured in the groundwater observation wells.

2.3.8.2 Salinization

In the salinization scenario, the MgZ follows a clear sequence. The MgZ value rises initially, then falls until it becomes negative (Figure 2.6). The samples could therefore be divided into three salinization stages: initial salinization with a positive MgZ (- in Figure 2.7), intermediate salinization with a neutral MgZ (~ in Figure 2.7), and late-stage salinization with a negative MgZ (— in Figure 2.7). When initial, intermediate and late-stage salinizing groundwater is found in sequence, the direction of salinization can be interpreted. Groundwater at the salinization front is in the initial stage, behind it is intermediate salinizing groundwater, and finally late-stage salinizing groundwater is close to the source of the saline water.

Two clear salinization sequences are observed in the study area. One is south of the central settlement, with late-stage salinization close to the aquaculture ponds followed by intermediate salinization and initial salinization towards the fresh groundwater under the central village (Figure 2.7). This sequence indicates that the aquifer has been salinized from the surface of the lower areas. Another salinization sequence is visible at a former creek near N4 and N5 (Figure 2.7). Although the clay at the very top is freshening due to recent freshwater recharge, late-stage salinization is visible in the shallow aquifer, followed by intermediate and initial salinization in the brackish and fresh samples at around 12 m depth (Figure 2.7). This indicates that some of the groundwater has been salinized by water infiltrating from this former creek.

Just north of the central settlement, the salinizing samples lack a clear sequence, but the different salinization stages can still be compared with each other to reveal differences in salinization processes. Under the agricultural fields near N1 and P1 the shallow brackish to fresh–brackish samples display initial salinization, whereas there is late-stage salinization in the samples from depths of 36.5 and 45.7 m. The samples from 36.5 m below the village on higher land again display initial salinization. This suggests that the deeper subsurface under the lower areas has been salinized for longer than the shallower subsurface.

The samples taken below the aquaculture ponds at P11, P12 and P14 display intermediate or late-stage salinization, suggesting that they have been salinized by water infiltrating from the aquaculture ponds above. However, as there are no samples close by with different salinization characteristics, it was not possible to determine the direction of this salinization front.

Some of the fresh and brackish water samples taken in the central settlement also display salinization. They have probably been salinized by some limited local saline water recharge, as the cation exchange characteristic in the fresh samples is sensitive to small changes in salinity. Near N3, the source of this local salinization is probably water infiltrating from the surrounding low areas, as the salinizing samples were taken close to the edge of the higher-lying area (Figure 2.7).

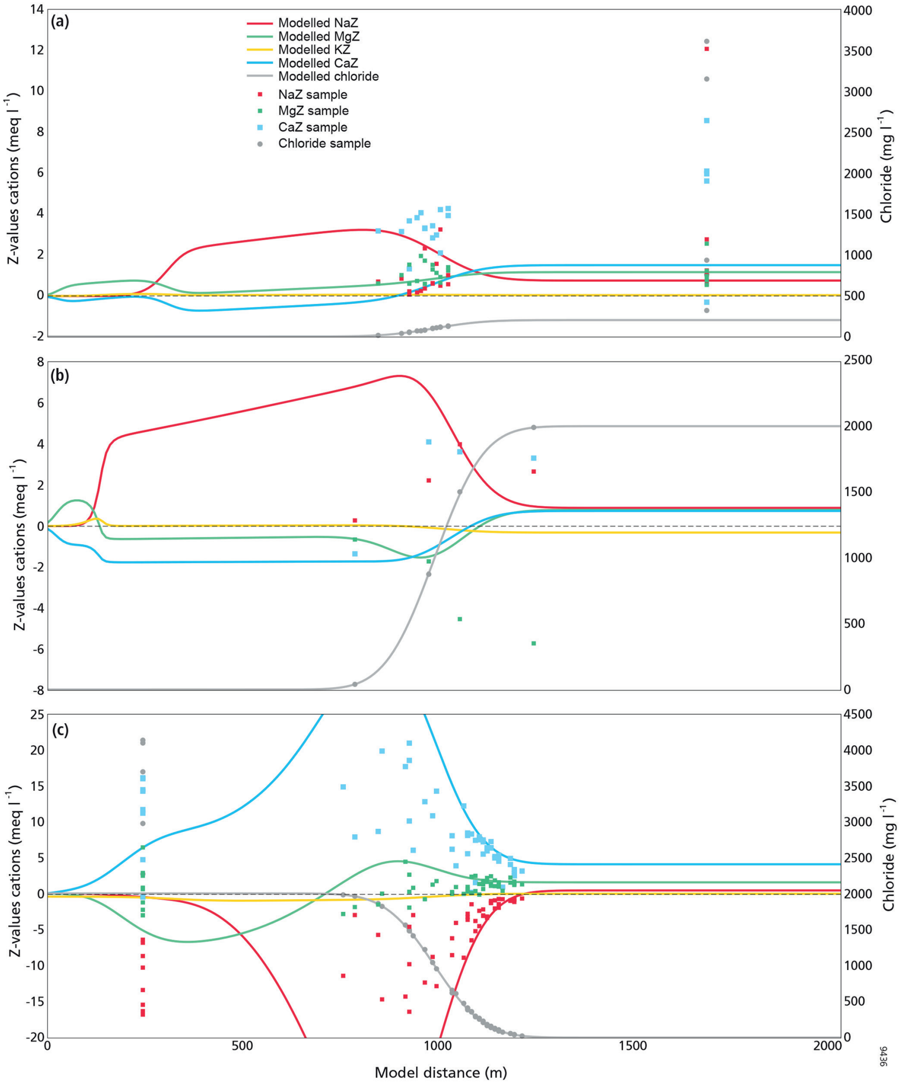


Figure 2.6. Results of the freshwater freshening scenario (A), the saline water freshening scenario (B) and the salinization scenario (C), together with the samples that match each scenario according to the Z-values of their cations. The X location of the samples is determined by the chloride concentration matching the values of the chloride in the scenario. Samples with a chloride concentration exceeding the chloride values in the scenario are plotted at x= 250 or x=1750.

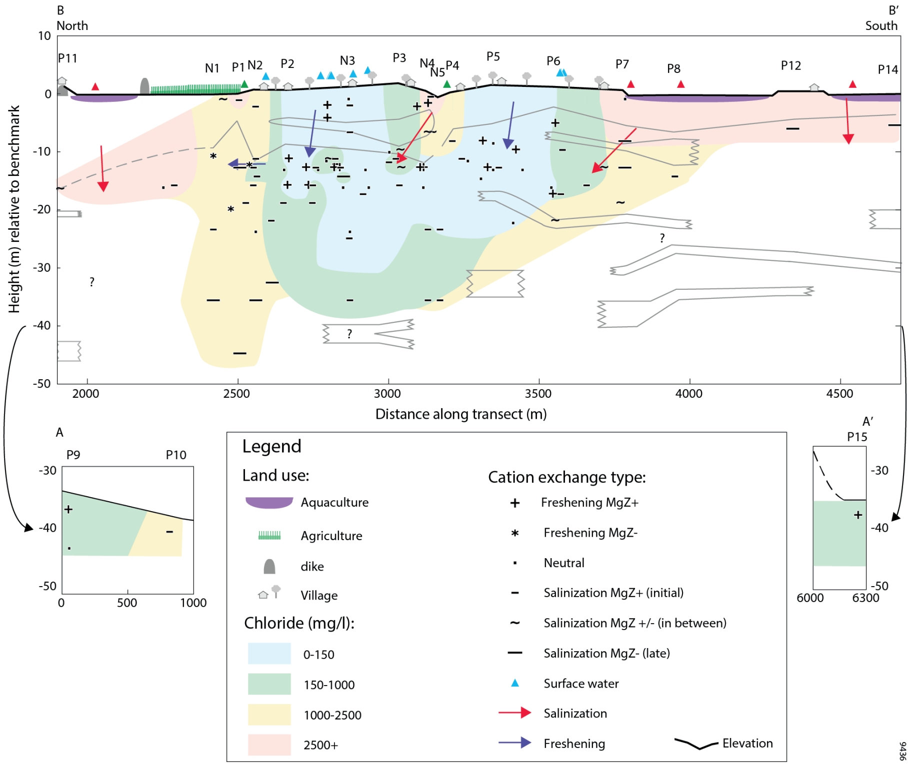


Figure 2.7. Cation exchange types of the sampled groundwater and the phreatic water from the surface clay layer

2.4 Discussion

2.4.1 Inferred hydrogeological evolution of the study area

Combining the field study results with the geological reconstruction based on literature enabled us to postulate a reconstruction of the evolution of the lithology and salinity distribution in the groundwater. This evolution is discussed below, considering the three main Holocene sedimentation phases described earlier plus an additional phase describing present-day processes.

2.4.1.1 Phase 1: Filling of an incised valley during Holocene transgression (10 kyr – 7kyr BP)

The Pleistocene palaeosol was not observed in the study area, which indicates either that a large incised valley was present in the Pleistocene, or that a

Holocene channel later truncated the palaeosol (Hoque et al. 2014, Goodbred et al. 2014, Sarkar et al. 2009). This incised valley or truncated Holocene channel filled up rapidly during the transgression, when a rapid sea level rise combined with an increase in monsoon intensity led to accelerated sedimentation of channel sands (Figure 2.8(a)) (Islam and Tooley, 1999; Goodbred and Kuehl, 2000a; Goodbred and Kuehl, 2000b; Sarkar et al., 2009; Ayers et al., 2016). Therefore, the deeper part of the first aquifer sands must have been during the transgression.

2.4.1.2 Phase 2: Emergence of large lithological differences (c. 7 kyr – 5/2.5 kyr BP)

At some point in the early Holocene, channel meandering and associated sand deposition became limited to the middle of the transect (Figure 2.8 (b)). This is indicated by the switch in sedimentary conditions from sand to clay deposition at approximately 30–35 m depth in the lower-lying areas at P9, P10 in the north, and P15 in the south. The unimodal, poorly sorted grain size distribution of this clay indicates that this switch probably occurred during the progradation (Sarkar et al., 2009). Throughout most of the rest of the Holocene, the areas near P9, P10 and P15 remained mangrove-forested tidal delta floodplains, while sand continued to be deposited by the Holocene channel around the present-day central village. The large difference in lithology indicates that the location of the channel was stable throughout the progradation. Possibly, mangrove vegetation in the floodplains prevented the channels from meandering due its ability to capture sediments up to the mean high water level (Furukawa and Wolanski, 1996; Auerbach et al., 2015) and to protect land against erosion (Van Santen et al., 2007; Kirwan and Megonigal, 2013).

This difference in lithology steered the influence of surface water on groundwater during the rest of the Holocene. The groundwater in the sandy aquifer in the middle of the transect continued to be influenced by the fresh surface water conditions, while the aquifers below the floodplains were much more isolated from surface influences by the thick clay layer. The groundwater under the thick clay layer must therefore be controlled by the hydrological conditions at the time of burial. Consequently, the thickness of the clay is the factor controlling the relative importance of palaeohydrological conditions for present-day groundwater salinity. We assume that during the progradation, the salinity at the surface decreased, as evidenced by the freshening cation exchange observed in the brackish groundwater near P9 and P15 (Figure 2.7). Additionally, the notion that this groundwater formed under different circumstances than the water close to the present-day central village is reinforced by the different isotopic composition of the groundwater below the thick clay layer compared to the isotopic compositions of the groundwater in the middle of the transect. This process of connate water sealing with subsequent limited influence has also been proposed by George (2013), Worland et al. (2015) and Ayers et al. (2016).

2.4.1.3 Phase 3: Clay deposition and formation of elevation differences (5/2.5 kyr – present)

a) Clay deposition

After the Ganges migrated eastwards between 5 kyr and 2.5 kyr BP, the areas that contained large sandy Holocene channels during the progradation also started to develop a clay cover (Figure 2.8(c)) (Goodbred and Kuehl, 2000a; Allison et al., 2003; Goodbred et al., 2003; Sarkar et al., 2009; Goodbred et al., 2014). The salinity during the deposition of the clay cover is not known, as it has been greatly affected by more recent freshening and salinization processes (see phase 4). Overall, however, more brackish conditions are likely to have prevailed from the moment the Ganges migrated eastwards, because the upstream supply of fresh water decreased. Possibly, the deeper groundwater at N1, P1 and N2 has been affected by salinization from this period, since its cation exchange characteristic indicates salinization at a later stage than the shallower samples (Figure 2.7).

Possibly, starting during the clay deposition in phase 3, salinization of the edges of the aquifer under the thick clay cover has occurred, due to density-driven flow from saline water infiltration in adjacent areas with a thin clay cover (Kooi et al., 2000). In the study area, however, brackish conditions are present at P16 and P17, indicating that density-driven flow has not affected the groundwater immediately below the thick clay layer. This does not mean that density-driven flow is not relevant in the study area. As salinization from density-driven flow mainly consists of vertical convection cells, the resulting salinization can be expected to occur deeper than immediately below the thick clay cover (Kooi et al., 2000; Smith and Turner, 2001).

b) Formation of elevation differences

While the clay cover was being deposited in phase 3, there was probably little difference in elevation. The observed differences in elevation are thought to have come about after the clay deposition, as a result of differences in autocompaction (Allen, 2000; Bird et al., 2004; Trönqvist et al., 2008), which can lead to an inversion of surface elevation (Vlam, 1942; van der Sluijs et al., 1965). The thick, organic-matter-rich clays under the floodplains are likely to have been compacted more than the thinner clay cover on the former sand channels, which would account for the present-day elevation differences of up to 1.5 m between the floodplains and the central former channel area in the village on higher land. These elevation differences are similar to those observed in a comparable delta area in the Netherlands (Vlam, 1942; van der Sluijs et al., 1965).

The elevation differences in the middle part of the transect cannot be explained by autocompaction, as here only small changes in lithology are observed. Instead, we hypothesize that they may result from erosion by creeks at the edge of the higher areas. Evidence for this is provided by landforms that look like pathways of erosion caused by meandering tidal creeks north of the central settlement, the tidal creek soils in the areas hypothesized to be affected by erosion, and the distribution of tidal creeks and former tidal creeks in lower-lying areas overlain by thin clay (FAO, 1959). Erosion by tidal creeks implies that the clays originally lay above the average tidal river water level. A possible explanation is clay deposition during the higher sea level between 4.5 kyr and 2 kyr BP (Gupta and Amin, 1974, Mathur et al. 2004, Sarkar et al. 2009). In a subsequent stage, the average water level of the nearby tidal creeks dropped again, resulting in erosive channels.

2.4.1.4 Phase 4: Emergence of groundwater salinity differences (present-day processes)

a) Higher areas: Freshening by rain and rainfed ponds

The present-day small differences in elevation result in large differences in groundwater salinity, as the surface elevation has determined whether freshening or salinization has occurred in the groundwater. In the higher-lying areas, the conditions at the surface have mostly been fresh since the elevation differences came about, as the slight elevation has prevented saline water flooding from tides and tidal surges. The fresh groundwater is recharged either by direct infiltration of rainwater, or by infiltration of rainwater stored in man-made ponds. Direct rainwater infiltration, which is a common formation process of freshwater lenses in elevated zones within saline areas (Goes et al., 2009; de Louw et al., 2011; Stuyfzand, 1993; Walraevens et al., 2007), is indicated by the light isotopic composition in the phreatic groundwater from the clay at N3.

Infiltration from the rainfed man-made ponds by humans could be a source of fresh groundwater, since they contain fresh water year-round (Harvey et al., 2006; Sengupta et al., 2008). This enables infiltration of fresh water in the dry season when the hydraulic head between the ponds and the groundwater is expected to be larger than in the wet season. The evaporated isotopic composition of the groundwater at P6 suggests such infiltration of pond water (Figure 2.5). However, the isotopic composition of the deeper fresh groundwater shows only a small amount of evaporation (Figure 2.4, Figure 2.5), indicating that infiltration from the freshwater ponds is not the main process responsible for the fresh groundwater – a conclusion reinforced by the usually very low permeability of the pond bottoms (Sengupta et al., 2008), and the fact that construction of freshwater ponds occurred relatively recently in geological terms (Kräzlin, 2000).

b) Low areas: salinization by marine-influenced water

Unlike the higher areas, the lower areas have often been flooded by tides or tidal surges. Recharge of this saline surface water has been possible in the lower areas with a thin clay cover, where salinizing saline groundwater was observed. This difference between salinization in the lower areas and freshening in the higher areas causes the surface elevation to be the most important factor controlling the salinity of the groundwater in areas with a thin clay cover. Since the salinization originates from the surface, there is already a large difference between higher and lower areas in terms of salinity in the uppermost part of the subsurface, similar to the finding reported by Fernández et al. (2010) for a delta area in Spain. This differs from the situation in Zeeland, the Netherlands, where small fresh groundwater lenses are also present in low-lying areas (Goes et al., 2009; de Louw et al., 2011).

Erosion by tidal creeks also occurred in the middle of the central settlement, as visible at the former creek location near N4, N5 and P4, which has salinized the existing fresh groundwater body, as evidenced by the chloride concentrations (Figure 2.3) and the sequential salinization patterns (Figure 2.7) recharging on top of the fresh groundwater.

Recently, aquifers under thin surface clay layers have become salinized by water infiltrating from the overlying aquaculture ponds. This is observed at P11, P7 and P14, where the groundwater has a similar salinity to the overlying aquaculture ponds and an isotopic composition that indicates large evaporation and mixing effects. The salinization by saline aquaculture was detected only in the shallow groundwater underneath saline aquaculture ponds in lower areas with thin surface clay layers. Slightly deeper, the salinization from the shrimp farms was no longer observed; the samples from approximately 12 m deep at P7 and P8 are already much less saline (respectively 1230 and 1370 mg/L), and a totally different isotopic composition

(Figure 2.3, Figure 2.6). Furthermore, no effect was observed in aquifers under thick clay layers. This rather limited influence of saline aquaculture is not unexpected, as saline aquaculture was only introduced in the study area approximately 30 years ago (Azad et al., 2009). It does indicate that land use has become a controlling factor for the shallow groundwater salinity in areas with thin surface clay layers. In the future, salinization from aquaculture ponds is expected to continue, and hence the extent of salinization to increase. Since the low-lying aquaculture areas with thin surface clay layers are adjacent to the higher areas, continued salinization from the aquaculture ponds could be a threat for the fresh groundwater under the higher area.

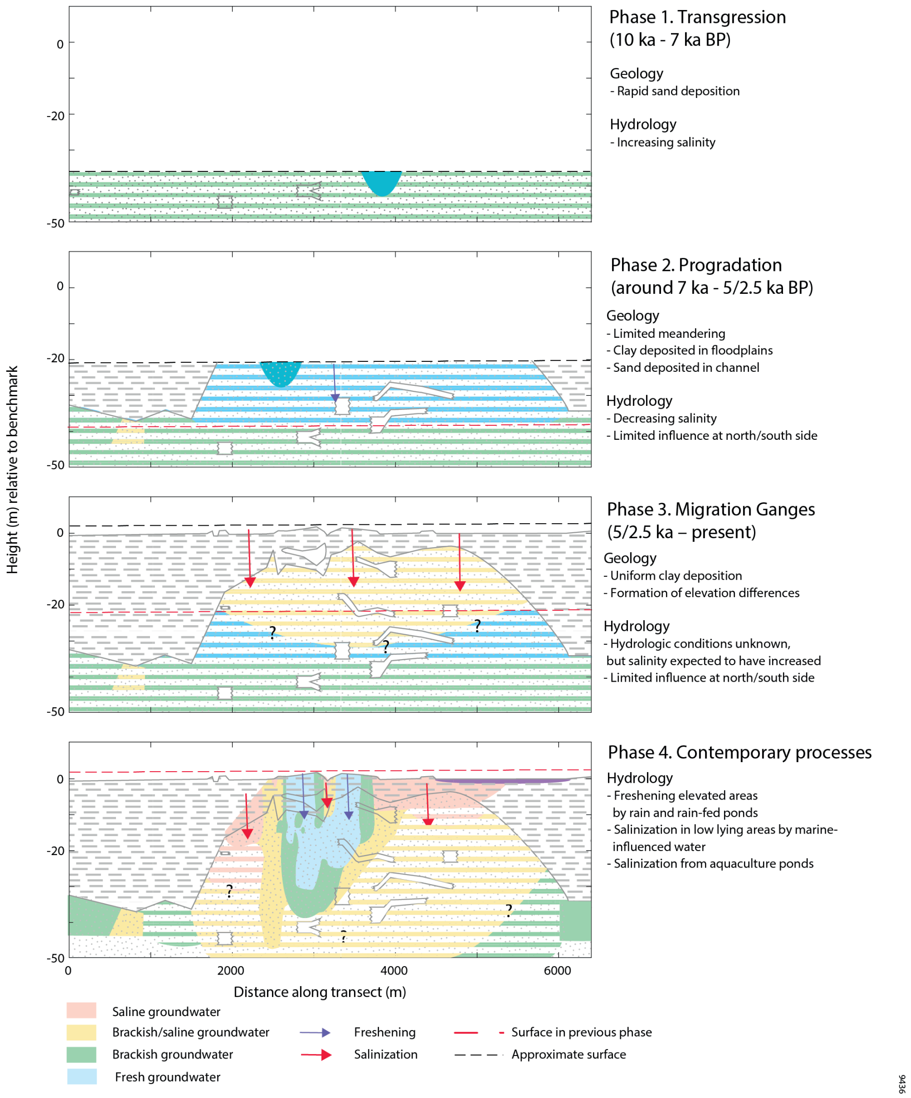


Figure 2.8. Sediment build-up and associated freshening and salinization processes in the study area.

2.4.2 Reflection

Above, we derived a hydrogeological evolution of a small area in southwestern Bangladesh, focusing on explaining the large variation in lithology and groundwater salinity which has often been reported in southwestern

Bangladesh (BGS and DHPE, 2001; George, 2013; Worland et al., 2015; Ayers et al., 2016). Thanks to the high density of the sampling and the combination of salinity data with isotopic data and PHREEQC-interpreted cation exchange data, it was possible to establish clear patterns in groundwater salinity and to identify relevant hydrological processes and geographical and geological controls. Under the slightly higher area with a thin surface clay layer, a clear pattern of fresh groundwater was identified, which is attributable to recharge by direct infiltration of rain or via rainfed ponds. The presence of such fresh groundwater lenses in this region was postulated by Worland et al. (2015) and Ayers et al. (2016), and the occurrence of fresh groundwater in elevated areas has been described in other brackish or saline deltas (Stuyfzand, 1993; Walraevens et al., 2007; Goes et al., 2009; de Louw et al., 2011; Santos et al., 2012), but to the best of our knowledge, these phenomena in southwestern Bangladesh have never previously been reported in such detail. The fresh groundwater is bordered by brackish and brackish–saline groundwater at greater depth under the higher area and in the direction of the adjacent lower areas, which probably indicates mixing of fresh groundwater with recharged saline flood waters from the lower areas. Saline groundwater is found only at relatively shallow depths below aquaculture ponds in areas with a thin surface clay layer and is attributed to recharge from these present-day aquaculture ponds. Under thick surface clay layers in the lower areas, brackish water is found; it is postulated to be controlled by palaeo salinity conditions at the time of sealing of the sand aquifer. The importance of palaeo conditions for groundwater salinity in isolated parts of the subsurface in this region has been mentioned by others (George, 2013; Worland et al., 2015; Ayers et al., 2016) and has been described in other coastal zones (Sukhija et al., 1996; Groen et al., 2000; Post and Kooi, 2003; Sivan et al., 2005). We postulate the hydrological processes described above and the resulting observed groundwater salinity variation to be primarily steered by three geological and geographical controlling factors: clay cover thickness, relative elevation and present-day land use.

We acknowledge our study has several limitations, and our interpretations should be seen as a conceptual model to explain the observed spatial patterns of clay and sand deposits and of groundwater salinity. We did not focus on quantifying recharge, discharge and flow rates, or the exact time scales of the hydrological processes. Without age dating, we can’t determine the exact moment of salinization or freshening that has occurred. Additionally, we have been unable to discern comprehensive groundwater flow directions, aside from sketching some indicative flow directions that would account for recharge, as we could not find evidence for locations and patterns of upward groundwater flow and discharge. These upward groundwater flows are expected to be present in convection cells caused by density-driven salinization (Kooi et al., 2000; Smith and Turner, 2001), and discharge is anticipated at drainage points in the landscape which are thought to be present at the edge of higher areas and at the lowest points in the landscape, i.e. the tidal rivers (Tóth, 1963). A possible next step would be to develop a numerical model to further elucidate these flow processes, as well as estimates of recharge and discharge rates and time scales of the described hydrological processes.

Despite these limitations, we contend that the identified controlling factors (clay cover thickness, relative elevation and present-day land use) satisfactorily explain an appreciable part of the observed variation in groundwater salinity variation in the larger southwestern Bangladesh region. Relative elevation and land use data could provide a first estimate of the groundwater salinity in areas with a thin surface clay layer, while knowledge of the palaeohydrogeological conditions seems to be necessary to understand and predict the groundwater salinity in areas with a thick surface clay layer. A next step would be to test the validity of this hypothesis at regional scale.

Acknowledgements

This work is part of the Delta-MAR project funded by the Urbanizing Deltas of the World programme of NWO-WOTRO. We would like to thank all staff from the Delta-MAR office in Khulna for all their support during the fieldwork, notably Abir Delwaruzzaman for his continued efforts to make sure the fieldwork campaigns were successful. From Dhaka University, Atikul Islam and Pavel Khan are thanked for their field assistance. MSc students Frank van Broekhoven and Rebecca van Weesep (Utrecht University), and Aria Hamann (TU Delft) are acknowledged for helping collect and make a first interpretation of the field data. Dr Joy Burrough is acknowledged for editing a near-final version of the manuscript and Ton Markus is acknowledged for editing the figures.

APPENDIX A

Table A2.1. The results of the sediment sample analyses. For the bold TGA values no clear peak was identified, indicating influence of TGA noise. For the italic TGA values the peak overlapped the border between the organic matter and the carbonate temperatures.italic

Sample number	Sample location	Depth (m)	Grain size percentage						Thermogravimetric analysis	
			% <8 μm	% 8–63 μm	% 63–126 μm	% 126–252 μm	% 252–502 μm	% >502 μm	% Organic matter (weight loss 150–550 °C)	% carbonates (weight loss 550–850 °C)
1	N1	5	15	57	21	5	2	1	1.51	2.43
2	N1	12	2	11	19	52	14	1	**0.49**	2.02
3	N1	24	0	7	18	51	23	2	**0.45**	2.01
4	N1	37	16	18	13	30	19	4	1.34	1.90
5	P1	5	40	56	2	0	0	2	*0.40*	*1.13*
6	P1	14	1	9	39	44	6	1	**0.33**	2.58
7	N2	5	14	63	18	4	1	0	1.26	2.79
8	N2	12	5	18	32	40	5	0	**0.57**	2.33
9	N2	24	1	10	15	48	19	7	**0.41**	2.04
10	N2	37	0	10	19	46	20	5	**0.55**	2.15
11	P2	8	6	28	33	24	6	2	**0.76**	2.69
12	P2	12	0	9	18	54	16	2	**0.56**	2.15
13	N3	5	41	56	2	0	0	2	*0.29*	*1.84*
14	N3	8	2	12	27	48	10	1	**0.58**	2.02
15	N3	11	15	49	17	14	4	2	2.81	2.57
16	N3	24	1	8	13	44	29	5	**0.30**	1.87
17	N3	37	0	6	8	32	40	14	**0.30**	1.96
18	P3	5	30	63	4	1	0	2	*0.51*	1.63
19	P3	9	0	9	29	44	14	3	**0.56**	2.23
20	P3	24	0	11	21	35	25	9	**1.05**	1.97
21	N4	3	22	68	8	1	0	1	0.98	3.23
22	N4	8	1	9	16	40	27	7	4.86	1.56

Sample number	Sample location	Depth (m)	Grain size percentage						Thermogravimetric analysis	
23	N4	9	11	46	22	14	6	1	1.75	3.37
24	N4	24	0	9	27	48	13	3	**0.39**	2.50
25	N4	37	0	6	15	50	25	3	**0.32**	2.14
26	N5	9	15	43	16	17	7	2	1.86	2.99
27	N5	12	0	2	5	30	55	9	**0.41**	1.33
28	N5	24	0	6	17	58	17	2	**0.36**	2.18
29	N5	37	0	6	15	44	26	8	**0.39**	2.14
30	P4	3	24	67	6	1	1	2	1.18	3.63
31	P4	9	0	10	22	48	17	3	**0.46**	1.70
32	P5	5	22	65	9	2	1	2	1.18	4.26
33	P5	9	2	16	26	38	16	3	**0.36**	1.82
34	P6	3	14	55	22	6	2	1	0.42	3.73
35	P6	6	0	5	20	49	21	4	**0.44**	1.97
36	P6	9	7	23	25	36	8	2	1.48	3.00
37	P6	23	0	7	16	52	24	2	**0.31**	1.82
38	P6	24	17	55	17	6	3	1	3.07	2.63
39	P7	9	2	25	44	20	7	3	**0.83**	2.89
40	P7	14	1	11	28	39	16	5	1.55	2.07
41	P8	5	12	53	26	7	2	0	1.56	3.07
42	P8	15	0	7	31	43	16	3	**0.54**	2.15
43	P8	37	1	5	7	33	49	5	4.65	1.50
44	P9	3	32	61	4	0	0	2	*1.94*	*1.73*
45	P9	15	26	65	7	1	1	1	3.03	3.53
46	P9	32	23	65	10	1	1	0	4.86	**1.65**
47	P9	37	1	13	39	39	6	2	**0.54**	2.92

REFERENCES

Allen, J. 2000. Morphodynamics of Holocene salt marshes: a review sketch from the Atlantic and Southern North Sea coasts of Europe, Quaternary Sci. Rev., 19, 1155–1231, https://doi.org/10.1016/S0277-3791(99)00034-7.

Allison, M. A. and Kepple, E. B., 2001. Modern sediment supply to the lower delta plain of the Ganges-Brahmaputra River in Bangladesh, Geo-Marine Lett., 21, 66–74, https://doi.org/10.1007/s003670100069.

Allison, M. A., Khan, S. R., Goodbred, S. L., and Kuehl, S. A., 2003. Stratigraphic evolution of the late Holocene Ganges– Brahmaputra lower delta plain, Sediment. Geol., 155, 317–342, https://doi.org/10.1016/S0037-0738(02)00185-9.

Appelo, C. A. J. and Postma, D., 2005. Geochemistry, groundwater and pollution, 2nd Edn., A. A. Balkema, Rotterdam.

Appelo, C. A. J. and Willemsen, A., 1987. Geochemical calculations and observations on salt water intrusions, I. A combined geochemical/minxing cell model, J. Hydrol., 94, 313–330, https://doi.org/10.1016/0022-1694(87)90058-8.

Auerbach, L. W., Goodbred Jr., S. L., Mondal, D. R., Wilson, C. A., Ahmed, K. R., Roy, K., Steckler, M. S., Small, C., Gilligan, J. M., and Ackerly, B. A., 2015. Flood risk of natural and embanked landscapes on the Ganges– Brahmaputra tidal delta plain, Nat. Clim. Change, 5, 153–157, https://doi.org/10.1038/nclimate2472.

Ayers, J. C., Goodbred, S., George, G., Fry, D., Benneyworth, L., Hornberger, G., Roy, K., Karim, M. R., and Akter, F., 2016. Sources of salinity and arsenic in groundwater in southwest Bangladesh, Geochem. Trans., 17, 4, https://doi.org/10.1186/s12932-016-0036-6.

Azad, A. K., Jensen, K. R., and Lin, C. K., 2009. Coastal Aquaculture Development in Bangladesh: Unsustainable and Sustainable Experiences, Environ. Manage., 44, 800–809, https://doi.org/10.1007/s00267-009-9356-y.

Bahar, M. M. and Reza, M. S., 2010. Hydrochemical characteristics and quality assessment of shallow groundwater in a coastal area of Southwest Bangladesh, Environ. Earth Sci., 61, 1065–1073, https://doi.org/10.1007/s12665-009-0427-4.

Beekman, H. E. and Appelo, C. A. J., 1991. Ion chromatography of freshand salt-water displacement: Laboratory experiments and multicomponent transport modelling, J. Contam. Hydrol., 7, 21–37, https://doi.org/10.1016/0169-7722(91)90036-Z.

BGS and DPHE, 2001. Arsenic contamination of groundwater in Bangladesh, BGS Technical Report WC/00/19, edited by: Kinniburgh, D. G. and Smedley, P. L., British Geological Survey, Keyworth.

Bhuiyan, M. J. A. N. and Dutta, D., 2012. Assessing impacts of sea level rise on river salinity in the Gorai river network, Bangladesh, Estuar. Coast. Shelf Sci., 96, 219–227, https://doi.org/10.1016/j.ecss.2011.11.005.

Bhuiyan, M. A. H., Rakib, M. A., Dampare, S. B., Ganyaglo, S., and Suzuki, S., 2011. Surface water quality assessment in the central part of Bangladesh using multivariate analysis, KSCE J. Civ. Eng., 15, 995–1003, https://doi.org/10.1007/s12205-011-1079-y.

Bird, M. I., Fifield, L. K., Chua, S., and Goh, B., 2004. Calculating Sediment Compaction for Radiocarbon Dating of Intertidal Sediments, Radiocarbon, 46, 421–435, https://

doi.org/10.1017/S0033822200039734.

Burgess, W. G., Hoque, M. A., Michael, H. A., Voss, C. I., Breit, G. N., and Ahmed, K. M., 2010. Vulnerability of deep groundwater in the Bengal Aquifer System to contamination by arsenic, Nat. Geosci., 3, 83–87, https://doi.org/10.1038/ngeo750.

Chowdhury, N. T., 2010. Water management in Bangladesh: an analytical review, Water Policy, 12, 32–51, https://doi.org/10.2166/wp.2009.112.

De Goffau, A., Van Leeuwen, T. C., Van den Ham, A., Doornewaard, G. J., and Fraters, B., 2012. Minerals Policy Monitoring Programme Report 2007–2010, Methods and Procedures, RIVM Report 680717018, National Institute for Public Health and the Environment, Bilthoven, the Netherlands

de Louw, P. G. B., Eeman, S., Siemon, B., Voortman, B. R., Gunnink, J., van Baaren, E. S., and Oude Essink, G. H. P., 2011. Shallow rainwater lenses in deltaic areas with saline seepage, Hydrol. Earth Syst. Sci., 15, 3659–3678, https://doi.org/10.5194/hess-15-3659-2011.

Delsman, J. R., Hu-A-Ng, K. R. M., Vos, P. C., De Louw, P. G. B., Oude Essink, G. H. P., Stuyfzand, P. J., and Bierkens, M. F. P., 2014. Paleo-modeling of coastal saltwater intrusion during the Holocene: An application to the Netherlands, Hydrol. Earth Syst. Sci., 18, 3891–3905, https://doi.org/10.5194/hess-18-3891-2014.

ESRI: "World imagery" [basemap], Scale Not Given, available at: http://www.arcgis.com/home/item.html?id=10df2279f9684e4a9f6a7f08febac2a9, last access: 11 June 2018.

Fakhruddin, S. H. M. and Rahman, J., 2014. Coping with coastal risk and vulnerabilities in Bangladesh, Int. J. Disast. Risk Reduct., 12, 112–118, https://doi.org/10.1016/j.ijdrr.2014.12.008.

FAO, 1959. Soil Survey of the Ganges-Kobadak Area, ETAP Report no. 1071, Rome.

Farr, T. G., Rosen, P. A., Caro, E., Crippen, R., Duren, R., Hensley, S., Kobrick, M., Paller, M., Rodriguez, E., Roth, L., Seal, D., Shaffer, S., Shimada, J., Umland, J., Werner, M., Oskin, M., Burbank, D., and Alsdorf, D., 2007. The Shuttle Radar Topography Mission, Rev. Geophys., 45, RG2004, https://doi.org/10.1029/2005RG000183.

Fernández, S., Santín, C., Marquínez, J., and Álvarez, M. A., 2010. Saltmarsh soil evolution after land reclamation in Atlantic estuaries (Bay of Biscay, North coast of Spain), Geomorphology, 114, 497–507, https://doi.org/10.1016/j.geomorph.2009.08.014.

Furukawa, K. and Wolanski, E., 1996. Sedimentation in mangrove forests, Mangrov. Salt Marsh., 1, 3–10, https://doi.org/10.1023/A:1025973426404.

George, G. J., 2013. Characterization of salinity sources in southwestern Bangladesh evaluated through surface water and groundwater geochemical analyses, MS Thesis, Vanderbilt University, Vanderbilt.

Goes, B. J. M., Oude Essink, G. H. P., Vernes, R. W., and Sergi, F., 2009. Estimating the depth of fresh and brackish groundwater in a predominantly saline region using geophysical and hydrological methods, Zeeland, the Netherlands, Near Surf. Geophys., 7, 401–412, https://doi.org/10.3997/1873-0604.2009048.

Goodbred, S. L. and Kuehl, S. A., 2000a. The significance of large sediment supply, active tectonism, and eustasy on margin sequence development: Late Quaternary stratigraphy and evolution of the Ganges–Brahmaputra delta, Sediment. Geol., 133, 227–248, https://doi.org/10.1016/S0037-0738(00)00041-5.

Goodbred, S. L. and Kuehl, S. A., 2000b. Enormous Ganges-Brahmaputra sediment discharge during strengthened early Holocene monsoon, Geology, 28, 1083, https://doi.org/10.1130/0091-7613(2000)28<1083:EGSDDS>2.0.CO;2.

Goodbred, S. L., Kuehl, S. A., Steckler, M. S., and Sarker, M. H., 2003. Controls on facies distribution and stratigraphic preservation in the Ganges–Brahmaputra delta sequence, Sediment. Geol., 155, 301–316, https://doi.org/10.1016/S0037-0738(02)00184-7.

Goodbred, S. L., Paolo, P. M., Ullah, M. S., Pate, R. D., Khan, S. R., Kuehl, S. A., Singh, S. K., and Rahaman, W., 2014. Piecing together the Ganges-Brahmaputra-Meghna River delta: Use of sediment provenance to reconstruct the history and interaction of multiple fluvial systems during Holocene delta evolution, Geol. Soc. Am. Bull., 126, 1495–1510, https://doi.org/10.1130/B30965.1.

Griffioen, J., 2003. Kation-uitwisselingspatronen bij zoet/zout grondwaterverplaatsingen, Stromingen, 9, 35–45.

Griffioen, J., 2017. Enhanced weathering of olivine in seawater: The efficiency as revealed by thermodynamic scenario analysis, Sci. Total Environ., 575, 536–544, https://doi.org/10.1016/j.scitotenv.2016.09.008.

Griffioen, J., Vermooten, S., and Janssen, G., 2013. Geochemical and palaeohydrological controls on the composition of shallow groundwater in the Netherlands, Appl. Geochem., 39, 129–149, https://doi.org/10.1016/j.apgeochem.2013.10.005.

Groen, J., Velstra, J., and Meesters, A. G. C., 2000. Salinization processes in paleowaters in coastal sediments of Suriname: evidence from 37Cl analysis and diffusion modelling, J. Hydrol., 234, 1–20, https://doi.org/10.1016/S0022-1694(00)00235-3.

Gupta, S. K. and Amin, B. S., 1974. Io/U ages of corals from Saurashtra coast, Mar. Geol., 16, M79–M83, https://doi.org/10.1016/0025-3227(74)90065-6.

Harvey, C. F., Swartz, C. H., Badruzzaman, A. B. M., Keon-Blute, N., Yu, W., Ali, M. A., Jay, J., Beckie, R., Niedan, V., Brabander, D., Oates, P. M., Ashfaque, K. N., Islam, S., Hemond, H. F., and Ahmed, M. F., 2002. Arsenic Mobility and Groundwater Extraction in Bangladesh, Science, 298, 1602–1606, https://doi.org/10.1126/science.1076978.

Harvey, C. F., Ashfaque, K. N., Yu,W., Badruzzaman, A. B. M., Ali, M. A., Oates, P. M., Michael, H. A., Neumann, R. B., Beckie, R., Islam, S., and Ahmed, M. F., 2006. Groundwater dynamics and arsenic contamination in Bangladesh, Chem. Geol., 228, 112–136, https://doi.org/10.1016/j.chemgeo.2005.11.025.

Hoogsteen, M. J. J., Lantinga, E. A., Bakker, E. J., Groot, J. C. J., and Tittonell, P. A., 2015. Estimating soil organic carbon through loss on ignition: effects of ignition conditions and structural water loss, Eur. J. Soil Sci., 66, 320–328, https://doi.org/10.1111/ejss.12224.

Hoque, M. A., McArthur, J. M., and Sikdar, P. K., 2014. Sources of low-arsenic groundwater in the Bengal Basin: investigating the influence of the last glacial maximum palaeosol using a 115-km traverse across Bangladesh, Hydrogeol. J., 22, 1535–1547, https://doi.org/10.1007/s10040-014-1139-8.

Horneman, A., van Geen, A., Kent, D. V., Mathe, P. E., Zheng, Y., Dhar, R. K., O'Connell, S., Hoque, M. A., Aziz, Z., Shamsudduha, M., Seddique, A. A., and Ahmed, K. M., 2004.

Decoupling of As and Fe release to Bangladesh groundwater under reducing conditions. Part I: Evidence from sediment profiles, Geochim. Cosmochim. Ac., 68, 3459–3473, https://doi.org/10.1016/j.gca.2004.01.026.

IAEA/WMO: Global Network of Isotopes in Precipitation, The GNIP Database, available at: http://www.iaea.org/water, last access: 26 September 2017.

Islam, M. S. and Tooley, M. J., 1999. Coastal and sea-level changes during the Holocene in Bangladesh, Quatern. Int., 55, 61–75, https://doi.org/10.1016/S1040-6182(98)00025-1.

Kakiuchi, H., Momoshima, N., Okai, T., and Maeda, Y., 1999. Tritium concentration in ocean, J. Radioanal. Nucl. Chem., 239, 523–526, https://doi.org/10.1007/BF02349062.

Khan, A. E., Scheelbeek, P. F. D., Shilpi, A. B., Chan, Q., Mojumder, S. K., Rahman, A., Haines, A., and Vineis, P., 2014. Salinity in Drinking Water and the Risk of (Pre) Eclampsia and Gestational Hypertension in Coastal Bangladesh: A Case-Control Study, edited by: Szecsi, P. B., PLoS One, 9, e108715, https://doi.org/10.1371/journal.pone.0108715.

Kirwan, M. L. and Megonigal, J. P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise, Nature, 504, 53–60, https://doi.org/10.1038/nature12856.

Konert, M. and Vandenberghe, J., 1997. Comparison of laser grain size analysis with pipette and sieve analysis: a solution for the underestimation of the clay fraction, Sedimentology, 44, 523–535, https://doi.org/10.1046/j.1365-3091.1997.d01-38.x.

Kooi, H., Groen, J., and Leijnse, A., 2000. Modes of seawater intrusion during transgressions, Water Resour. Res., 36, 3581–3589, https://doi.org/10.1029/2000WR900243.

Kränzlin, I., 2000. Pond management in rural Bangladesh: problems and possibilities in the context of the water supply crisis, Nat. Resour. Forum, 24, 211–223, https://doi.org/10.1111/j.1477-8947.2000.tb00945.x.

Mathur, U. B., Pandey, D. K., and Bahadur, T., 2004. Falling Late Holocene sea-level along the Indian coast, Curr. Sci., 87, 439–440.

Morgan, J. P. and McIntire, W. G., 1959. Quarternary Geology of the Bengal Basin, East Pakistan and India, Geol. Soc. Am. Bull., 70, 319–342, https://doi.org/10.1130/0016-7606(1959)70[319:QGOTBB]2.0.CO;2.

Mukherjee, A., Fryar, A. E., and Thomas, W. A., 2009. Geologic, geomorphic and hydrologic framework and evolution of the Bengal basin, India and Bangladesh, J. Asian Earth Sci., 34, 227–244, https://doi.org/10.1016/j.jseaes.2008.05.011.

Parkhurst, D. L. and Appelo, C. A. J., 2013. Description of Input and Examples for PHREEQC Version 3 – A Computer Program for Speciation, Batch-Reaction, One-Dimensional Transport, and Inverse Geochemical Calculations, in: Techniques and Methods, book 6, chap. A43, US Geological Survey, Denver, p. 497, available at https://pubs.usgs.gov/tm/06/a43/ (last access: March 2019).

Paul, B. G. and Vogl, C. R., 2011. Impacts of shrimp farming in Bangladesh: Challenges and alternatives, Ocean Coast. Manage., 54, 201–211, https://doi.org/10.1016/j.ocecoaman.2010.12.001.

Post, V. E. A. and Kooi, H., 2003. Rates of salinization by free convection in high-permeability sediments: insights from numerical modelling and application to the Dutch coastal area, Hydrogeol. J., 11, 549–559, https://doi.org/10.1007/s10040-003-0271-7.

Rahman, M. M., Hassan, M. Q., Islam, M. S., and Shamsad, S. Z. K. M., 2000. Environmental impact assessment on water quality deterioration caused by the decreased Ganges outflow and saline water intrusion in south-western Bangladesh, Environ. Geol., 40, 31–40, https://doi.org/10.1007/s002540000152.

Ravenscroft, P., Burgess, W. G., Ahmed, K. M., Burren, M., and Perrin, J., 2005. Arsenic in groundwater of the Bengal Basin, Bangladesh: Distribution, field relations, and hydrogeological setting, Hydrogeol. J., 13, 727–751, https://doi.org/10.1007/s10040-003-0314-0.

Rezaei, M., Sanz, E., Raeisi, E., Ayora, C., Vázquez-Suñé, E., and Carrera, J., 2005. Reactive transport modeling of calcite dissolution in the fresh-salt water mixing zone, J. Hydrol., 311, 282–298, https://doi.org/10.1016/j.jhydrol.2004.12.017.

Santos, I. R., Eyre, B. D., and Huettel, M., 2012. The driving forces of porewater and groundwater flow in permeable coastal sediments: A review, Estuar. Coast. Shelf Sci., 98, 1–15, https://doi.org/10.1016/j.ecss.2011.10.024.

Sarkar, A., Sengupta, S., McArthur, J. M., Ravenscroft, P., Bera, M. K., Bhushan, R., Samanta, A., and Agrawal, S., 2009. Evolution of Ganges–Brahmaputra western delta plain: Clues from sedimentology and carbon isotopes, Quaternary Sci. Rev., 28, 2564–2581, https://doi.org/10.1016/j.quascirev.2009.05.016.

Sarin, M. M., Krishnaswami, S., Dilli, K., Somayajulu, B. L. K., and Moore, W. S., 1989. Major ion chemistry of the Ganga–Brahmapura river system:Weathering processes and fluxes to the Bay of Bengal, Geochim. Cosmochim. Ac., 53, 997–1009.

Sengupta, S., McArthur, J. M., Sarkar, A., Leng, M. J., Ravenscroft, P., Howarth, R. J., and Banerjee, D. M., 2008. Do Ponds Cause Arsenic-Pollution of Groundwater in the Bengal Basin? An Answer from West Bengal, Environ. Sci. Technol., 42, 5156–5164, https://doi.org/10.1021/es702988m.

Shameem, M. I. M., Momtaz, S., and Rauscher, R., 2014. Vulnerability of rural livelihoods to multiple stressors: A case study from the southwest coastal region of Bangladesh, Ocean Coast. Manage., 102, 79–87, https://doi.org/10.1016/j.ocecoaman.2014.09.002.

Shamsudduha, M. and Uddin, A., 2007. Quaternary shoreline shifting and hydrogeologic influence on the distribution of groundwater arsenic in aquifers of the Bengal Basin, J. Asian Earth Sci., 31, 177–194, https://doi.org/10.1016/j.jseaes.2007.07.001.

Sharma, B., Amarasinghe, U., Xueliang, C., de Condappa, D., Shah, T., Mukherji, A., Bharati, L., Ambili, G., Qureshi, A., Pant, D., Xenarios, S., Singh, R., and Smakhtin, V., 2010. The indus and the ganges: River basins under extreme pressure,Water Int., 35, 493–521, https://doi.org/10.1080/02508060.2010.512996.

Sivan, O., Yechieli, Y., Herut, B., and Lazar, B., 2005. Geochemical evolution and timescale of seawater intrusion into the coastal aquifer of Israel, Geochim. Cosmochim. Ac., 69, 579–592, https://doi.org/10.1016/j.gca.2004.07.023.

Smith, A. J. and Turner, J. V., 2001. Density-dependent surface water-groundwater interaction and nutrient discharge in the Swan-Canning Estuary, Hydrol. Process., 15, 2595–2616, https://doi.org/10.1002/hyp.303.

Steefel, C. I. and Macquarrie, K. T. B., 1996. Approaches to modeling of reactive transport in porous media, Rev. Mineral. Geochem., 34, 85–129.

Stuyfzand, P. J., 1993. Hydrochemistry and hydrology of the coastal dune area of the Western Netherlands, PhD thesis, Vrije Universiteit of Amsterdam, Amsterdam.

Sukhija, B. S., Varma, V. N., Nagabhushanam, P., and Reddy, D.V., 1996. Differentiation of palaeomarine and modern seawater intruded salinities in coastal groundwaters (of Karaikal and Tanjavur, India) based on inorganic chemistry, organic biomarker fingerprints and radiocarbon dating, J. Hydrol., 174, 173–201, https://doi.org/10.1016/0022-1694(95)02712-2.

Törnqvist, T. E., Wallace, D. J., Storms, J. E. A., Wallinga, J., van Dam, R. L., Blaauw, M., Derksen, M. S., Klerks, C. J. W., Meijneken, C., and Snijders, E. M. A., 2008. Mississippi Delta subsidence primarily caused by compaction of Holocene strata, Nat. Geosci., 1, 173–176, https://doi.org/10.1038/ngeo129.

Tóth, J., 1963. A theoretical analysis of groundwater flow in small drainage basins, J. Geophys. Res., 68, 4795–4812, https://doi.org/10.1029/JZ068i016p04795.

Umitsu, M., 1993. Late quaternary sedimentary environments and landforms in the Ganges Delta, Sediment. Geol., 83, 177–186, https://doi.org/10.1016/0037-0738(93)90011-S.

van der Molen, W. H., 1958. The exchangeable cations in soils flooded with sea water, Wageningen University, Wageningen.

van der Sluijs, P., Steur, G. G. L., and Ovaa, I., 1965. DE BODEM VAN ZEELAND, toelichting bij blad 7 van de Bodemkaart van Nederland, schaal 1 : 200 000, Centrum voor Landbouwpublikaties en Landbouwdocumentatie, Wageningen.

van Gaans, P. F. M., Griffioen, J., Mol, G., and Klaver, G., 2011.Geochemical reactivity of subsurface sediments as potential buffer to anthropogenic inputs: a strategy for regional characterization in the Netherlands, J. Soils Sediments, 11, 336–351, https://doi.org/10.1007/s11368-010-0313-4.

Van Santen, P., Augustinus, P. G. E. F., Janssen-Stelder, B. M., Quartel, S., and Tri, N. H., 2007. Sedimentation in an estuarine mangrove system, J. Asian Earth Sci., 29, 566–575, https://doi.org/10.1016/j.jseaes.2006.05.011.

Vlam, A. W., 1942. Historisch-morfologisch onderzoek van eenige Zeeuwsche eilanden, PhD thesis, Utrecht Universiteit, Utrecht.

Walraevens, K., Cardenal-Escarcena, J., and Van Camp, M., 2007. Reaction transport modelling of a freshening aquifer (Tertiary Ledo-Paniselian Aquifer, Flanders-Belgium), Appl. Geochem., 22, 289–305, https://doi.org/10.1016/j.apgeochem.2006.09.006.

Worland, S. C., Hornberger, G. M., and Goodbred, S. L., 2015. Source, transport, and evolution of saline groundwater in a shallow Holocene aquifer on the tidal deltaplain of southwest Bangladesh, Water Resour. Res., 51, 5791–5805, https://doi.org/10.1002/2014WR016262.

CHAPTER 3

INFLUENCE OF LANDSCAPE FEATURES ON THE LARGE VARIATION OF SHALLOW GROUNDWATER SALINITY IN SOUTHWESTERN BANGLADESH

Floris Loys Naus

Paul Schot

Kazi Matin Ahmed

Jasper Griffioen

2019. Journal of Hydrology X 5,100043

ABSTRACT

In southwestern Bangladesh, the large variation in groundwater salinity has only been elucidated in small-scale study areas and along large regional-scale gradients. We aimed to assess the regional shallow (<60 m) groundwater salinity variation with a higher resolution as a function of landscape features and associated hydrological processes. Spatial variation in groundwater salinity was assessed using 442 EC measurements from previous studies and 1998 new EC measurements. Groundwater EC values were correlated with well location data (latitude, longitude and depth of the filter) and landscape feature data (elevation, soil type, land use and surface clay thickness). Additionally, we performed a geomorphological analysis of landscape features to infer associated hydrological processes. We interpret wide fluvial zones to be remnants of sandy deposits in large paleo channels which allow freshwater recharge, resulting in groundwater that is mostly (75%) fresh. Narrow fluvial zones, tidal fluvial zones, and fluvial zones next to tidal rivers are more susceptible to lateral saline water flow or saline water recharge by occasional tidal flooding, and only contain some shallow fresh groundwater in high-lying zones. Tidal flat or tidal fringe zones hardly contain any fresh groundwater. This study is the first to demonstrate the relation between landscape features, hydrological processes and regional groundwater salinity throughout southwestern Bangladesh. The main lines of our approach may be applicable in other coastal areas with available spatial landscape feature data, enabling a first prediction of groundwater salinity variation.

3.1 INTRODUCTION

Coastal regions and deltas are among the most heavily populated areas in the world and their water resources are experiencing increasing stress (Creel, 2003; Ranjan et al., 2006; Nicholls and Cazenave, 2010). One of the largest and most densely populated deltas is the Ganges–Brahmaputra–Meghna (GBM) delta. Here, manifestations of this stress are arsenic contamination of shallow groundwater resources (Nickson et al., 1998; Harvey et al., 2002), severe pollution of surface water resources (Alam et al., 2006; Bhuiyan et al., 2011), and limited availability of the meteorological water resources, due to pronounced seasonality (Chowdhury, 2010; Sharma et al., 2010). In the coastal southwestern region of Bangladesh, available drinking water is further limited by the salinity of the surface- and groundwater. The groundwater salinity variation in the coastal area is large and, therefore, hard to predict (George, 2013; Worland et al., 2015; Ayers et al., 2016; Rahman et al., 2018; Naus et al., 2019). The stress of saline water intrusion on the groundwater is increasing due to natural changes in the form of natural land subsidence and sea level rise (Bhuiyan and Dutta, 2012; Fakhruddin and Rahman, 2014), and due to anthropogenic changes in the form of man-induced land subsidence, decreased Ganges outflow (Rahman et al., 2000) and increased groundwater extraction (Chowdhury, 2010). To determine occurrence of current drinking water problems and to mitigate future risk for the drinking water supply in southwestern Bangladesh, a proper understanding of the current groundwater salinity distribution and underlying controlling processes is required. Such general understanding can also aid in understanding groundwater salinity variation in other coastal regions around the world.

Most previous studies in southwestern Bangladesh focussed on understanding the groundwater salinity variation by identifying the controlling hydrological processes in small, local study areas (George, 2013; Worland et al., 2015; Ayers et al., 2016; Sarker et al., 2018; Naus et al., 2019). These studies have found much of the groundwater to be connate groundwater due to the subsurface lithology being characterized by thick covering clay layers and the landscape being flat leading to a stagnant hydrological system. The occurrence and thickness of surface clay layers with low infiltration capacity is described to explain how isolated the groundwater is and, therefore, whether the groundwater consists of solely connate water formed under paleo conditions or whether the groundwater salinity has also been influenced by active processes (George, 2013; Worland et al., 2015; Ayers et al., 2016; Sarker et al., 2018; Naus et al., 2019). Previous studies proposed several active processes that vary depending on the present-day landscape features. Freshwater recharge from rain or rain-fed pond water has been described in areas of higher elevation in southwestern Bangladesh (Naus et al., 2019), as well as in other coastal regions (Stuyfzand, 1993; Walraevens et al., 2007; Goes et al., 2009; de Louw et al., 2011; Santos et al., 2012). Salinization in low-lying areas has been attributed to three processes in southwestern Bangladesh: tidally influenced flooding (Naus et al., 2019), recharge by rainwater that has dissolved evaporite salts in the soil (Sarker et al., 2018), and recharge from aquaculture ponds (Paul and Vogl, 2011; Rahman et al., 2018; Naus et al., 2019). At a regional scale, the relevance of these hydrological processes is unexplored and, accordingly, the regional groundwater salinity variation remains poorly understood: Only some rough regional salinity gradients have been delineated, indicating a trend for salinity to increase from north to south (Bangladesh Water Development Board, 2013; Zahid et al., 2018). For assessing and overcoming water supply problems in the region a spatially more detailed understanding of the current groundwater salinity distribution and underlying controlling processes is required. We, therefore, set out to elucidate the regional shallow (<60 m deep) groundwater salinity variation with a higher resolution and to determine the relevant controlling hydrological processes in a regional-scale study. We aimed to use our findings to construct guidelines for predicting the groundwater salinity throughout the region.

3.2 STUDY AREA

The study region (latitude 22° 12' to 23° 0', longitude 88° 53' to 89° 58') comprises the three southwesternmost districts of Bangladesh: Satkhira, Khulna and Bagerhat. The southern parts of these districts contain the largest tidal mangrove forest in the world, the Sundarbans, which was excluded from the study area. The remaining area is predominantly rural, with the main sources of income being agriculture and aquaculture. The topography, soil type and land use vary throughout the study area (Figures A3.1, A3.2, A3.3). Most of the area is low-lying with an elevation a few meters above mean sea level (Auerbach et al., 2015), increasing regionally towards the north and east (Figure A3.1). At a smaller scale the topography is more varied (Figure A3.1), with some high-lying areas up to 2 m higher than the low-lying areas (Naus et al., 2019). The dominant land use types are treed villages, one-season rice, aquaculture and irrigated rice, with treed villages and irrigated rice being located more often towards the north and at higher elevations than aquaculture or one-season rice (Naus et al., 2019; Figure A3.2). A soil map is available for the Satkhira and Khulna districts from the Food and Agriculture

Organization of the United Nations (FAO) (Food and Agriculture Organization of the United Nations, 1959, Figure A3.3), indicating a variety of fluvial, tidal fluvial, tidal flat and tidal fringe soils, with fluvial soils occurring more in the north and in the west, tidal fringe soils occurring more in the east, and tidal flat soils occurring more in the south. The fluvial soils often occur on higher land; the other soil types also sometimes occur on relatively high-lying areas.

The region experiences an annual rainfall of around 2500 mm, with most of this precipitation falling in the monsoon from June to October. The monsoon season is followed by a cool dry winter in October to March, and a hot summer from March to June. The tidal rivers and creeks become saline at the end of the summer, in April and May (Bhuiyan and Dutta, 2012). In general, three aquifer types are recognized in Bangladesh (Mukherjee et al., 2009). In the south of Bangladesh, the Upper Holocene aquifers are found to depths of approximately 60 m. Below are the main, Middle Holocene aquifers, which are found to approximately 150 m deep; the Late Pleistocene–Early Holocene aquifers extend deeper than 150 m. This study focusses on the groundwater salinity in the Upper Holocene aquifers, i.e. the first 60 m.

3.3 METHODS

3.3.1 General approach

We used a three-way approach. First, we assessed how groundwater salinity throughout the region correlates with landscape features. Second, we performed a geomorphological analysis of landscape features to infer associated hydrological processes and their effect on groundwater salinity. Third, using landscape features we constructed practical guidelines to predict groundwater salinity throughout the region.

For our approach we required data already available or that could be rapidly collected at the scale of southwestern Bangladesh. We used three types of data: (1) Electrical Conductivity (EC) data from previous studies and collected during this study, to determine groundwater salinity; (2) Location data (x,y,z) of the EC measurements, to analyse how EC varies with depth and throughout the region; and (3) Geomorphological landscape features from various existing large-scale spatial data sources.

3.3.2 Data collection, data mining and data processing

The EC data was collected by the authors and MSc students from Dhaka University during multiple fieldwork campaigns from January 2014 to September 2018, using a low-cost method that consisted of measuring the EC of shallow groundwater (<60 m deep) by pumping household tube wells along regional transects and detailed study areas throughout the region. The protocol for measuring EC from the tube wells included the purging of the stagnant water from the tube well by pumping each tube well for at least a minute. We expect the measurements collected over the span of several years to be comparable, as seasonal and inter-annual salinity variation is expected to be low because of the stagnancy of the connate-groundwater-dominated hydrological system (George, 2013; Worland et al., 2015; Ayers et al., 2016; Sarker et al., 2018; Naus et al., 2019). This low-cost method made it possible to collect 1998 new EC groundwater measurements at tube wells. Three handheld field measurement devices were used for the EC measurements: the HANNA HI 9829 (Hanna 127 Instruments, USA), the Hanna HI 98,311 (Hanna 127 Instruments, USA), and the 18.50.SA multimeter (Eijkelkamp, The Netherlands). Additional chemical analysis of the water samples was performed for a subset of sampling locations. For this, 221 groundwater samples were taken along 10 local-scale transects in January 2017 and January 2018. The samples were filtered through a 0.45 μm membrane and stored in 15 ml polyethylene tubes for analysis in the lab for Cl, SO_4 and NO_3 using ion chromatography. Alkalinity was measured within 36 h of sampling by titration (Hach Company, USA).

This data was augmented with data on shallow groundwater (<60 m) from previously published articles and projects and from a datafile obtained from Vanderbilt University, see Table 3.1 for details. In total, our database consisted of 2440 EC measurements from the Upper Holocene aquifers (Table 3.1). The regional database on groundwater analyses was augmented with data from Naus et al. (2019) and Ayers et al. (2016), which provided an additional 136 samples, leading to a total of 357 hydrochemical samples (Table 3.1). These samples were used to infer the salinity from EC measurements, as explained in *Section 3.3.3*. The database of groundwater EC, chloride and alkalinity is available as Supplementary data.

Table 3.1. Groundwater and lithological data sources used in the study

Data source	EC measurements of the shallow aquifer (< 60m deep)	Samples with chloride and HCO3 data	Lithological data
This study	1998	221	14
Naus et al. (2019):	101	101	20
Ayers et al. (2016):	35	35	13
Data obtained from Vanderbilt University:	211	-	-
Bangladesh Water Development Board (BWDB, 2013):	3	-	110
British Geological Survey (BGS) and Department of Public Health Engineering (DPHE) (2001)	-	-	547
Unicef MAR project:	92	-	280
Total:	2440	357	984

Location data was collected for each EC measurement point during the field campaigns, and consisted of latitude, longitude and depth of the filter. The latitude and longitude of the measured tube wells were obtained by means of GPS, and the depth was estimated from details supplied by local residents on the filter depth and the number of pipes and filters used when installing the tube well. When the users sporadically did not remember or were in doubt about the depth of their tubewell, we did not include their tubewell.

For the landscape features, we used the data sources available at the scale of southwestern Bangladesh. These yielded data on elevation, land use, soil type and surface clay thickness. Elevation was estimated using Shuttle Radar Topography Mission data (SRTM) (Farr et al., 2007), taking into account that the exact values are not accurate as they have been affected by vegetation (Auerbach et al., 2015). Land use was based on supervised classification of cloudless Imagery from Landsat 8 (17th and 24th of March 2015, paths 137 and 138, row 44), calibrated with observations from aerial images and field observations. Soil type was derived from the FAO soil map (Food and Agriculture Organization of the United Nations, 1959). Since the soil map is approximately 70 years old, we could expect the dynamic nature of the delta to have led to differences compared to present-day circumstances, certainly on a small scale. However, visual similarities of the soil map (Figure A3.3) with the SRTM (Figure A3.1) and the land use map (Figure A3.2) give confidence in its present-day validity. The thickness of the surface clay layer was based on borehole logs from various previous studies and from 14 newly constructed tube wells, see Table 3.1 for details. As the variation in surface clay thickness in southwest Bangladesh is large (Ayers et al., 2016; Naus et al., 2019) the interpolated clay thickness is only used for EC measurements less than 100 m from a borehole log. There were 313 such EC measurements.

3.3.3 Definition of salinity

Most salinity classifications for drinking water are based on chloride concentration in the water (Custodio and Bruggeman, 1987; Stuyfzand, 1993). However, salinity and EC are not only controlled by Cl but also by other anions, particularly SO_4 and NO_3, and by alkalinity. Therefore, we first assessed to what extent chloride controls EC as well as salinity in the study region by comparing the available hydrochemical analyses against the EC values. This revealed a large spread in combinations of chloride concentrations and EC, which particularly affects the interpretation of the salinity in the lower ranges (Figure 3.1). At EC values lower than 5 mS/cm, alkalinity is prominent and can lead to an overestimated chloride concentration based on EC. Between 5 and 10 mS/cm, alkalinity is less prominent but may still be relevant when inferring salinity from the EC instead of from the chloride content. This confirms that EC is not always a good estimator for the chloride concentration. The large spread makes it impossible to perform chemical calculations based on the EC, such as calculations of the fractions of seawater. Nevertheless, the R^2 of 0.84 between EC and chloride gives us confidence to use EC to judge the groundwater salinity.

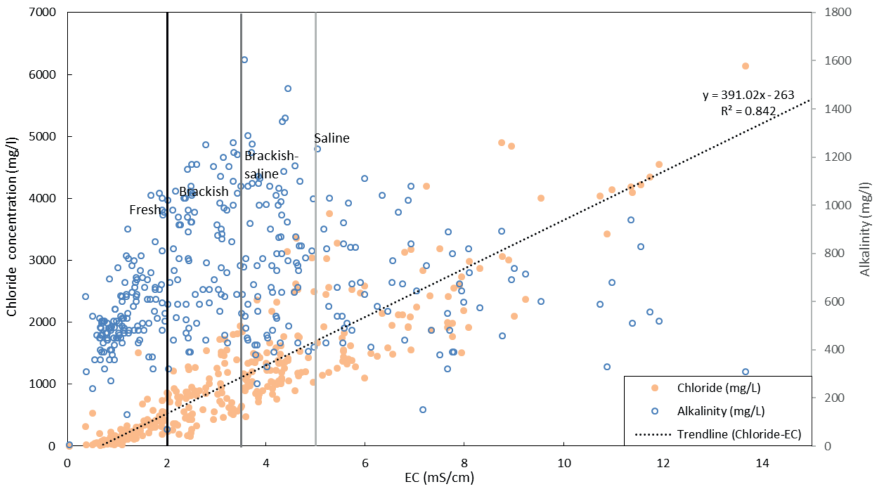


Figure 3.1. EC plotted against chloride concentration and alkalinity for the dataset of hydrochemical samples. The vertical lines indicate the boundaries between the four recognized salinity classes.

To differentiate between fresh and brackish groundwater, we used the drinking water guidelines of the Bangladesh government (Ayers et al., 2016), according to which groundwater with an EC of 2 mS/cm or less is potable. This definition is broader than the classification of Stuyfzand (1993), as it results in some of the measurements interpreted as fresh having a chloride concentration above 300 mg/l. The other salinity classes were constructed based on the Stuyfzand classification (1993) and the approximate correlation between EC and chloride (Figure 3.1). Groundwater was classified as brackish if it had an EC of 2–3.5 mS/cm, largely corresponding with a chloride concentration of 300–1000 mg/l. It was classified as brackish–saline if the EC was 3.5–5 mS/cm, corresponding with a chloride concentration 1000–2500 mg/l. It was classified as saline if the EC was > 5 mS/cm, corresponding with a chloride concentration > 2500 mg/l.

3.3.4 Statistical analysis

Various statistical and geostatistical tests were used to assess regional trends and EC variation with landscape features. Variation of EC with the ordinal location and landscape feature data (latitude, longitude, elevation, depth and surface clay thickness) was statistically analysed using Pearson's statistical test, with a significance threshold set at 0.05. Variation of EC with the nominal data consisting of land use and soil type was assessed using boxplots of the different classes. The distributions of the classes with 30 or more samples were tested for significant differences using Kruskal–Wallis (Kruskal and Wallis, 1952) and Dunn's tests (Dunn, 1964), with the significance threshold value adjusted according to Bonferroni correction (Dunn, 1961) by dividing the original value of 0.05 by the number of simultaneously tested classes. Regional trends in EC were also visually assessed after ordinary Kriging interpolation of the EC data, which has been applied before for soil salinity (Bilgili, 2013; Lobell et al., 2010). After trial and error, we decided to use a spherical variogram model, using the 8 nearest measurements to estimate the EC to prevent inclusion of data from too far away and to prevent oversensitivity to individual measurements.

To ascertain which variable controls the EC independently from possible cross-correlation, the EC data was split into classes according to latitude, longitude, elevation, clay thickness, depth, land use and soil type. The ordinal data was not assigned to these classes on the basis of cut-off points. Instead, expert judgement was used to obtain decently sized classes given the variable spread and number of measurements. The EC data was split into four subregions (northwest, northeast, southwest and southeast), three SRTM elevation classes (low, SRTM < 5 m; moderately high, 5 m ≤ SRTM < 8 m; and high, SRTM ≥ 8 m), two depth classes (shallow, depth < 30 m; deep, depth ≥ 30 m), and two clay thickness classes (thin, <10 m; thick ≥ 10 m). For the nominal data, the EC data was split into the different land use and soil classes. For each of these classes, the variation between the EC and the ordinal data was statistically analysed using Pearson's statistical test, with a significance threshold set at 0.05. For the soil classes, the patterns of EC with SRTM elevation and filter depth were additionally analysed following local regression (LOESS) which constructs a moving-window correlation using localized subsets of the data (Cleveland, 1979).

3.4 RESULTS

3.4.1 Regional variation

The Kriging interpolation reveals large-scale regional trends in groundwater EC (Figure 3.2). In general, EC values increase from north to south towards the sea, with most groundwater in the north being fresh and most groundwater in the south, close to the Sundarbans, being saline. Between the predominately fresh north and the saline south, the groundwater salinity variation is large. In Satkhira district (west), this variation is visible as alternating areas of saline and fresh groundwater; in the Khulna district (centre), small areas with fresher groundwater were detected between more saline areas, and in the Bagerhat district (east), the groundwater salinity varies the most in the eastern part.

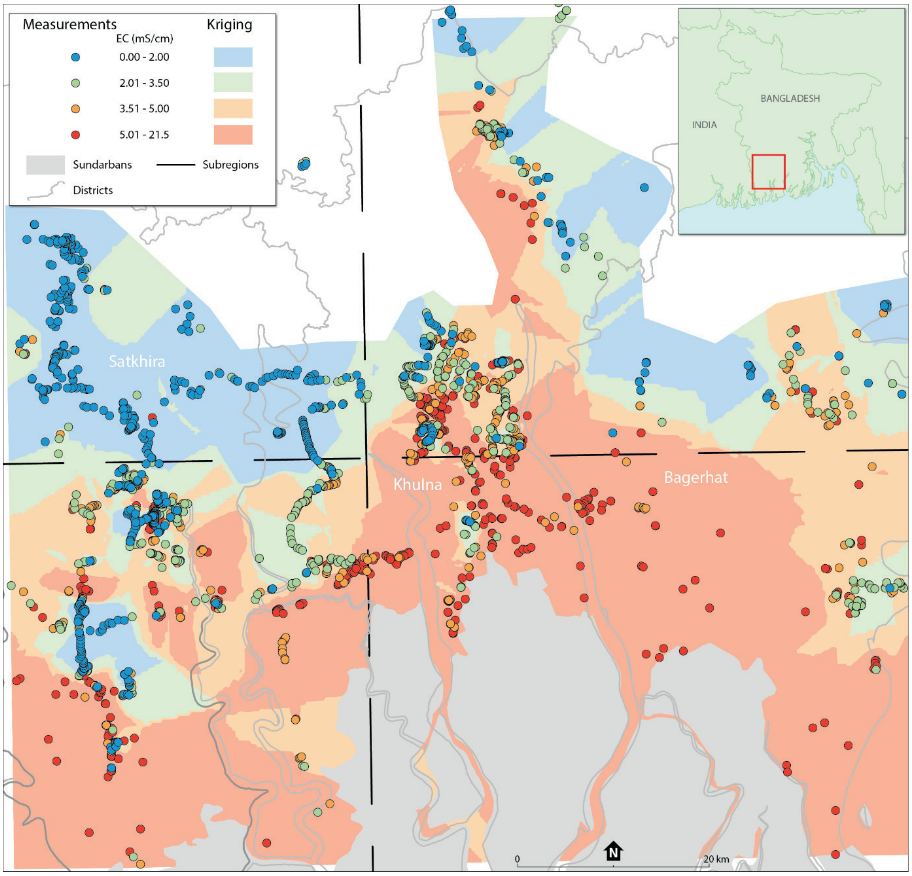


Figure 3.2. EC measurements from groundwater wells in the Upper Holocene aquifers (< 60 m depth), indicated with circles, and the associated Kriging interpolation in the background. The lines indicating the subregions separate the region into the classes 'Northwest', 'Northeast', 'Southwest' and 'Southeast', as used in Table 3.3. Note that the Kriging interpolation should be seen as a rough visual assistance for the regional variation in groundwater salinity. For site-specific groundwater salinity assessments, we refer to the guidelines in the discussion.

In Khulna district, where the Sundarbans extend furthest north, the area with predominately saline water extends further north than in the Satkhira and Bagerhat districts. The more saline nature of groundwater in Khulna compared to that in Satkhira and Bagerhat is also indicated by the absence of an area with predominately fresh groundwater: saline groundwater was detected up to the northern edge of Khulna district (Figure 3.2).

3.4.2 Landscape feature data and EC

Table 3.2 shows the significant correlations between EC, latitude, longitude, SRTM elevation, clay thickness and filter depth. The Pearson's correlation coefficients (|r|) are not high: they vary between 0.05 and 0.44. EC correlates significantly with latitude, longitude and SRTM elevation, but has no significant correlation with filter depth or clay thickness. The correlations with latitude and longitude affirm the trends visible in the Kriging interpolation: EC increases southwards and eastwards. The negative correlation with SRTM elevation indicates that groundwater is fresher under higher areas than under lower areas.

Table 3.2. Correlation matrix of the nominal data, p=0.05. Empty cells indicate correlations that are not significant. As a visual aid, we highlighted values between 0.3 and 0.4 purple, values between 0.4 and 0.5 light blue, and values above 0.5 green.

Latitude	-0.28				
Longitude	0.27	0.17			
Depth (m)		0.23	-0.05		
SRTM (m)	-0.25	0.17	0.15	-0.21	
Clay thickness (m)				0.44	-0.3
	EC (mS/cm)	Latitude	Longitude	Depth (m)	SRTM (m)

The EC boxplots of the land use classes (Figure 3.3) and the soil classes (Figure 3.4) reveal that there are significant differences in the salinity of the groundwater below the different land use and soil classes. For the land use classes, it should be noted that the sampling method has a bias in the number of measurements per land use class: most samples were taken in the treed villages, but the smallest group still contains 57 samples. Even so, significant differences are visible. Groundwater under irrigated rice is significantly fresher than groundwater under the other land use classes. Additionally, the one-season rice class contains significantly more saline groundwater than the treed villages class. The EC values for the irrigated rice class show rather limited salinity variation, while the salinity variation in the other land use classes is large (Figure 3.3).

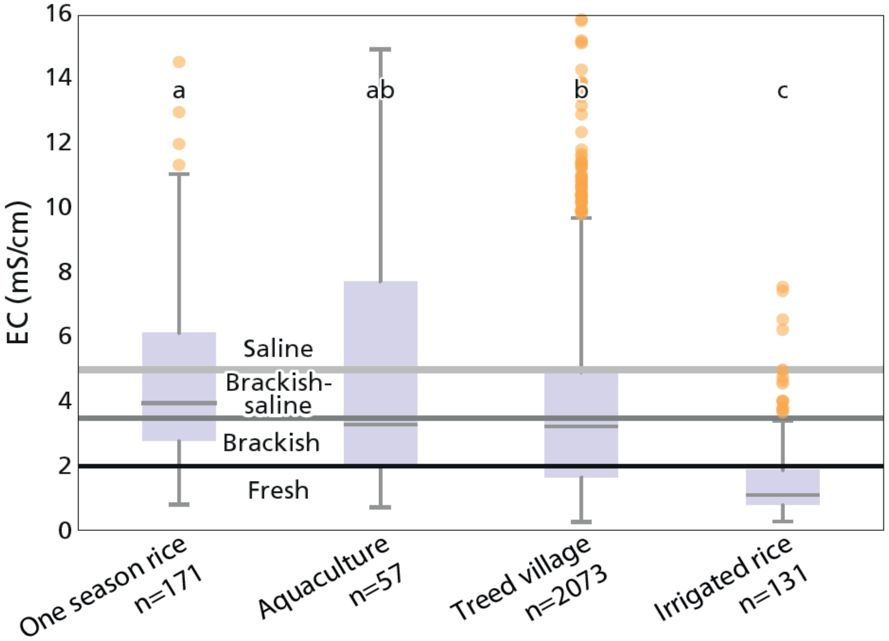


Figure 3.3. Boxplots of groundwater EC values for the different land use classes. The horizontal lines indicate the salinity classes and the letters above the boxes indicate whether the medians of the classes are significantly different based on Dunn's test (p= 0.008, following Bonferroni p-correction). Classes with similar letters have non-significant differences.

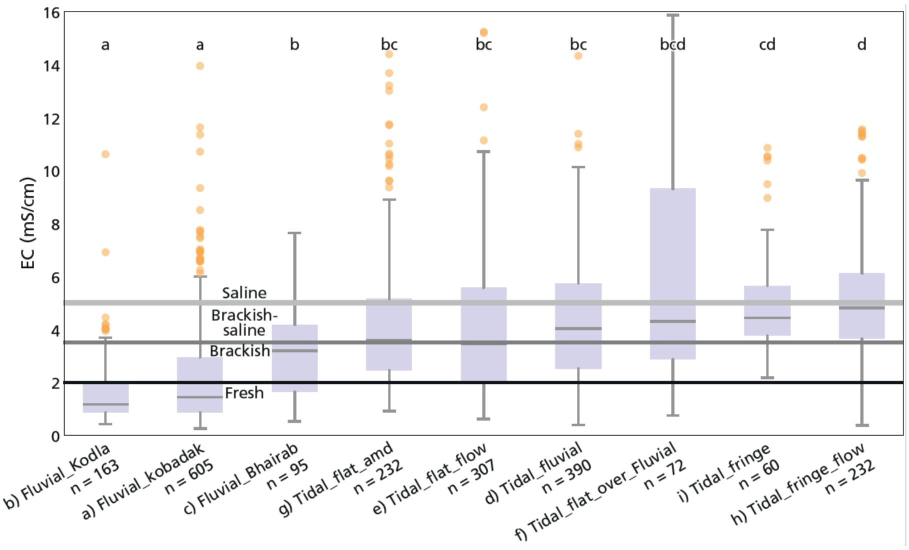


Figure 3.4. Boxplots of groundwater EC values for the different soil classes from the FAO soil map (1959), Figure A.3. The horizontal lines indicate the salinity classes and the letters above the boxes indicate whether the medians of the classes are significantly different based on Dunn's test (p= 0.0014, following Bonferroni p-correction). Classes with similar letters have non-significant differences.

The EC variance in each soil class is high: for each soil class, the interquartile range is in two or more salinity classes. Six soil classes were excluded from the significance testing because they had <30 observations. The Kobadak and Kodla fluvial soil classes (a, b) contain significantly fresher groundwater than the other soil classes (Figure 3.4), while the tidal fringe flow soil class contains significantly more saline groundwater than the following four soil classes: fluvial Bhairab (c), tidal flat flow (d), tidal flat Amd (e) and tidal fluvial (f). Lastly, groundwater under fluvial Bhairab soils is significantly fresher than under tidal fringe soils.

3.4.3 Other correlations

Significant correlations are visible between the location and landscape feature data of the measurements (Table 3.2). Some of these correlations are similar to the trends described above in Section 3.2. SRTM elevation is relatively high in the north, in the east, and for fluvial soils, treed villages and irrigated rice.

The correlations with depth and clay thickness were not described in Section 3.2. The clay thickness, SRTM elevation and filter depth are all intercorrelated. Measurements in wells with a deeper filter tend to be from locations with a thicker clay cap ($r = 0.44$) and a lower SRTM elevation ($r = -0.21$), and consequently clay caps are thicker in areas with a lower SRTM elevation (−0.3). There is also a trend for filters to be deeper towards the north ($r = 0.23$); the filter depths for the soil classes fluvial Bhairab, tidal flat flow and tidal flat Amd are significantly deeper than those for the other soil classes, while the filter depths are shallowest for irrigated rice and the deepest for aquaculture.

3.4.4 EC correlations of the split data

The correlations between EC and the ordinal data for each subclass of location or landscape feature data are presented in Table 3.3. In general, the correlation coefficients are higher within the subclasses of the independent variables than for the entire dataset. The EC correlations in the different subregions differ only slightly. There are differences in correlations with latitude and longitude among the subregions, but they reflect the trends visible in the Kriging map (Figure 3.2). There are no large differences in the EC correlations with SRTM elevation. Even in the northwest, where the groundwater under high and low areas is fresh, EC still varies with SRTM. This emphasizes the importance of this parameter on the groundwater EC. EC does, however, correlate differently with filter depth between the regions, with the southwest showing a slightly positive correlation ($r = 0.11$) and the northeast showing a slightly negative correlation (r = −0.11), while the northwest and southeast show no correlation.

Table 3.3. Pearson's correlation coefficients (r) for EC vs latitude, EC vs longitude, EC vs SRTM elevation, EC vs filter depth and EC vs clay thickness after the EC database was split in classes according to different parameters. Empty cells indicate correlations that are not significant. As a visual aid, we highlighted values between 0.3 and 0.4 purple, values between 0.4 and 0.5 light blue, and values above 0.5 green. Grey cells have fewer than 30 measurements. The order of the soil classes is given according to the average EC, with the letters in front of the soil classes being determined according to the original legend of the soil map (Figure A.3.3 and Figure 3.6).

	Median EC	n	EC vs lat.	EC vs long.	EC vs SRTM	EC vs Depth (m)	EC vs clay thickness
All	3.19	2438	-0.28	0.27	-0.25		
Northwest:	1.05	379			-0.28	-0.11	
Northeast:	3.75	856	-0.22	-0.14	-0.24		
Southwest:	2.94	907	-0.32	0.15	-0.31	0.11	
Southeast:	5.56	296		-0.2	-0.25		
Low (<5m)	3.62	532	-0.37	0.19		-0.29	-0.37
Mid (>=5, <8m)	3.58	1233	-0.27	0.35	-0.16		
High (>=8m)	1.9	673	-0.17	0.38			
Shallow (<30m)	3.02	1190	-0.25	0.22	-0.33		
Deep (>=30m)	3.35	1248	-0.34	0.37	-0.18		
Thin (<10m)	2.83	196	-0.32	0.24	-0.4	0.2	
Thick (>=10m)	4.43	117	-0.41	0.3			-0.23
b.) Fluvial Kodla:	1.18	163	-0.26		-0.31	-0.31	-0.54
a.) Fluvial Kobadak:	1.44	605	-0.19	0.17	-0.28	0.13	
c.) Fluvial Bhairab:	3.19	95		-0.21	-0.31	-0.34	
g.) Tidal flat Amd:	3.6	232	-0.49	-0.15		-0.29	
e.) Tidal flat flow:	3.45	307	-0.49	0.46		-0.37	
d.) Tidal fluvial:	4.04	390	-0.3	0.12	-0.16	0.18	
f.) Tidal flat on fluvial:	4.3	72	-0.54		-0.3		
i.) Tidal fringe:	4.44	60	-0.54	0.44			
h.) Tidal fringe flow:	4.81	232	-0.38	0.19	-0.22		
Treed village:	3.24	2073	-0.26	0.25	-0.26		-0.6
Aquaculture:	3.29	57	-0.5	0.36	0.28	-0.54	
One-season rice:	3.96	171	-0.4			-0.23	
Irrigated rice:	1.1	131					

Between the high, moderately high and low-lying areas, there are differences in the correlation of EC with latitude, longitude, SRTM elevation and filter depth (Table 3.3). In low-lying areas (SRTM elevation < 5 m), EC increases more strongly southwards and with filter depth, indicating that groundwater in low-lying areas in the south is more saline, and that shallow groundwater is more saline than deeper groundwater. Moderately high (SRTM elevation 5 – 8 m) and high areas (SRTM elevation ≥ 8 m) show a stronger increase in groundwater salinity towards the east, indicating that at higher elevations groundwater is more saline in the east than in the west. Only moderately high areas have a correlation with SRTM elevation, indicating that the change from fresher to more saline groundwater occurs gradually for the areas at intermediate elevations.

When split according to filter depth and clay thickness, clear differences in correlation are visible. The EC of the shallow groundwater and groundwater under a thin clay layer is controlled more strongly by elevation (Table 3.2), suggesting that the salinity of shallow or less isolated groundwater salinity is mostly controlled by elevation and not by regional-scale trends. In turn, deeper groundwater or groundwater under a thick clay layer correlates more strongly with latitude and longitude (Table 3.2), which indicates that the salinity of the more isolated groundwater is more related to regional salinity gradients.

There are differences in EC correlations between the different land use classes, when interpreting these differences, the uneven distribution of measurements across the land use classes must be taken into account. As most measurements are available for the treed village, the EC correlations are similar to those for the full dataset (Table 3.2). The correlations with EC for the land use classes aquaculture and one-season rice are similar to those for the low-lying areas: EC decreases with filter depth and also southwards, suggesting that saline water recharge from the surface occurs more often in the south and that it mostly affects the shallow groundwater.

Among the soil classes, there are different correlations with EC for latitude, elevation, depth and clay thickness (Table 3.2). The EC correlates more strongly with latitude in the tidal flat and tidal fringe soils than in the fluvial soils. The weaker correlation with latitude for the fluvial soils indicates that the fluvial Kodla and Kobadak soil classes contain fresh groundwater relatively independently from how far south the soils are located. The explanation for the absence of a regional EC gradient for the fluvial Bhairab soil class is that this soil type is present in only one location in the region, the north of Khulna district (Figure A3.3).

SRTM elevation is relevant for the groundwater salinity in the three fluvial soil classes and in the tidal flat on fluvial soil class, and to a lesser extent in the tidal fluvial and the tidal fringe flow soil classes (Table 3.3, Figure 3.5a). In the fluvial Kobadak and Kodla soils, the groundwater EC is mostly fresh in areas with a SRTM elevation above 6 m (Figure 3.5a), while in the tidal fluvial and fluvial Bhairab classes, measurements start to be mostly fresh at SRTM elevations above 10 m. In the tidal flat on fluvial soil class, a correlation with high salinities is found for the low SRTM elevations (SRTM elevation < 5 m, Figure 3.5a). EC does not correlate much with SRTM elevation for the tidal fringe, the tidal flat flow or the tidal flat Amd soil classes, which do not have many measurements at SRTM elevations above 10 m.

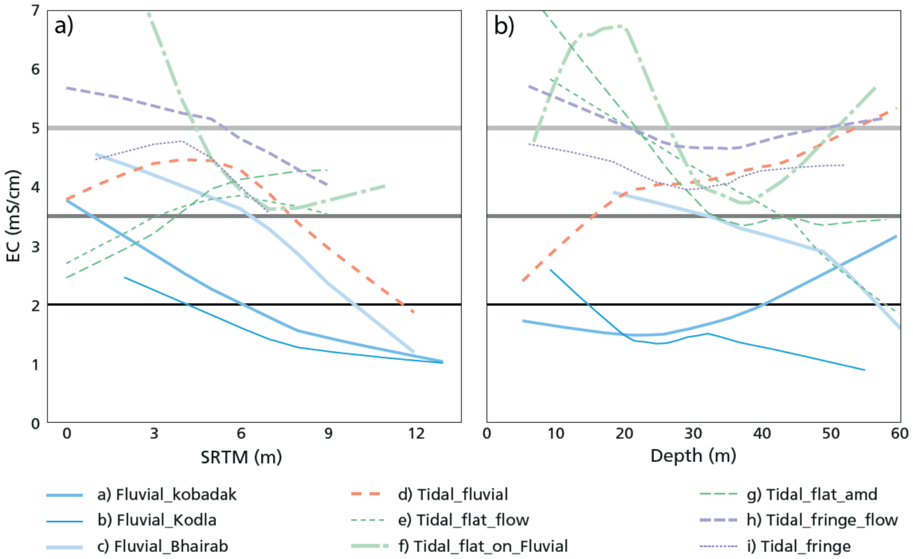


Figure 3.5. Local regressions (LOESS) of (a) EC vs SRTM and (b) EC vs depth (m) for the different soil classes. SRTM is specified only for round numbers, so a relatively high smoothing/fraction value of 0.9 was used to create the LOESS, indicating that 90% of the data was used to create each point. For the depth, a smoothing/fraction value of 0.6 was used.

Longitude correlates more strongly with EC for the tidal fringe and tidal flat flow soils than for the other soil types. For most soil classes, the salinity decreases with depth (Figure 3.5b). It decreases until approximately 30 m depth in the soil classes tidal fringe, tidal fringe flow, tidal flat on fluvial and tidal flat Amd, and it continues to drop until a depth of 60 m in the soil classes tidal flat flow, fluvial Kodla and fluvial Bhairab (Table 3.3, Figure 3.5b). Conversely, the salinity of the shallow groundwater is fresher than the salinity of the deeper groundwater in the fluvial Kobadak and tidal fluvial soil classes (Table 3.3, Figure 3.5b).

3.5 DISCUSSION

The results indicate that the groundwater EC varies regionally with SRTM elevation, and between soil classes and land use classes. However, the correlations are not definite, and do not explain all the variation. Nevertheless, general trends are revealed, which we discuss below. Subsequently, we present a geomorphological analysis of landscape features and we use the general trends as a basis to infer the associated hydrological processes to shed more light on what controls the groundwater salinity. The landscape features and their affiliated hydrological processes are then used to construct practical guidelines for predicting groundwater salinity throughout the region.

3.5.1 General trends

The correlation analysis and boxplots show some general trends between the landscape feature data and groundwater salinity. The groundwater under low-lying areas, one-season rice and tidal fringe soils is generally saline. The negative correlation of EC with filter depth in low-lying areas (SRTM elevation < 5 m), in areas with aquaculture, and in areas with one-season rice suggests the higher salinity there is caused by saline water recharge from the surface, as described in previous studies (Sarker et al., 2018; Paul and Vogl, 2011; Rahman et al., 2018; Naus et al., 2019). Additionally, the negative correlation of EC with clay thickness for the low-lying areas also suggests saline water recharge from the surface.

Under high-lying areas (SRTM elevation ≥ 8 m), under irrigated rice and under fluvial Kodla and Kodabak soils, groundwater is generally fresh. The high SRTM elevation contributes to recharge by precipitation that results in fresh groundwater, as described in previous studies in Bangladesh and elsewhere (Naus et al., 2019; Stuyfzand, 1993; Walraevens et al., 2007; Goes et al., 2009; de Louw et al., 2011; Santos et al., 2012). The Kobadak and Kodla soil classes reveal the occurrence of fresh groundwater most clearly, with the groundwater being significantly fresher than in the other soil classes (Figure 3.4). A high SRTM elevation, however, provides less guarantee for fresh groundwater, as there are also many measurements that indicate brackish, brackish–saline or saline groundwater at high SRTM elevations (Figure 3.5a). In the next section, we will discuss the relation between groundwater EC and landscape features in more detail.

3.5.2 Geomorphological analysis of landscape features

We used soil class as a primary indicator for landscape features and the geomorphological analysis as it is a reflection of sedimentology, elevation and operational hydrological processes. We clustered the soil classes in three landscape feature groups: (1) fluvial soils present in prominent large-scale meandering patterns, (2) tidal fluvial soils which surround the fluvial soils or are present in prominent smaller-scale meandering patterns, and (3) tidal flat and tidal fringe soils located in the low-lying flats or as moderately high to high narrow ridges (SRTM elevation 5– to 10 m) next to channels, creeks or roads.

3.5.2.1 Fluvial soils

The spatial occurrences of the fluvial Kodla, fluvial Kobadak and fluvial Bhairab soils resemble large-scale fossil meanders oriented from northwest to southeast throughout the region (Figure A3.3, Figure 3.6a, b). A detailed hydrogeological study of one of the fluvial Kobadak soils indicated that they consist of the sandy remnants of paleo channel deposits (Naus et al., 2019), with the elevation difference being caused by differences in autocompaction between the sandy sediments of the paleo channel and the clayey sediments of the surrounding paleo flats (Vlam, 1942; van der Sluijs et al., 1965; Naus et al., 2019). The high-lying areas with fluvial soils are expected to have formed similarly throughout the region. Here, the direction of the fluvial fossil meanders corresponds with the direction of meandering rivers during the progradation (Shamsudduha and Uddin, 2007). The large amounts of fresh groundwater found down to approximately 30 m under many of these fluvial soils could result from recharge of groundwater during the long period that the fossil meanders have been present, enhanced by the presence of very permeable sandy deposits (Vacher, 1988) and the absence of flooding with saline water from tides or tidal surges (Naus et al., 2019). Similar occurrence of fresh water recharge in higher elevated areas have been described in other brackish or saline coastal areas (Stuyfzand, 1993; Walraevens et al., 2007; Goes et al., 2009; de Louw et al., 2011; Santos et al., 2012). It should be noted that the areas with fresh water recharge are also more sensitive to waste water influences, such as from pit latrines in the treed villages, which could pollute and contribute some salinity to the groundwater (McArthur et al., 2012).

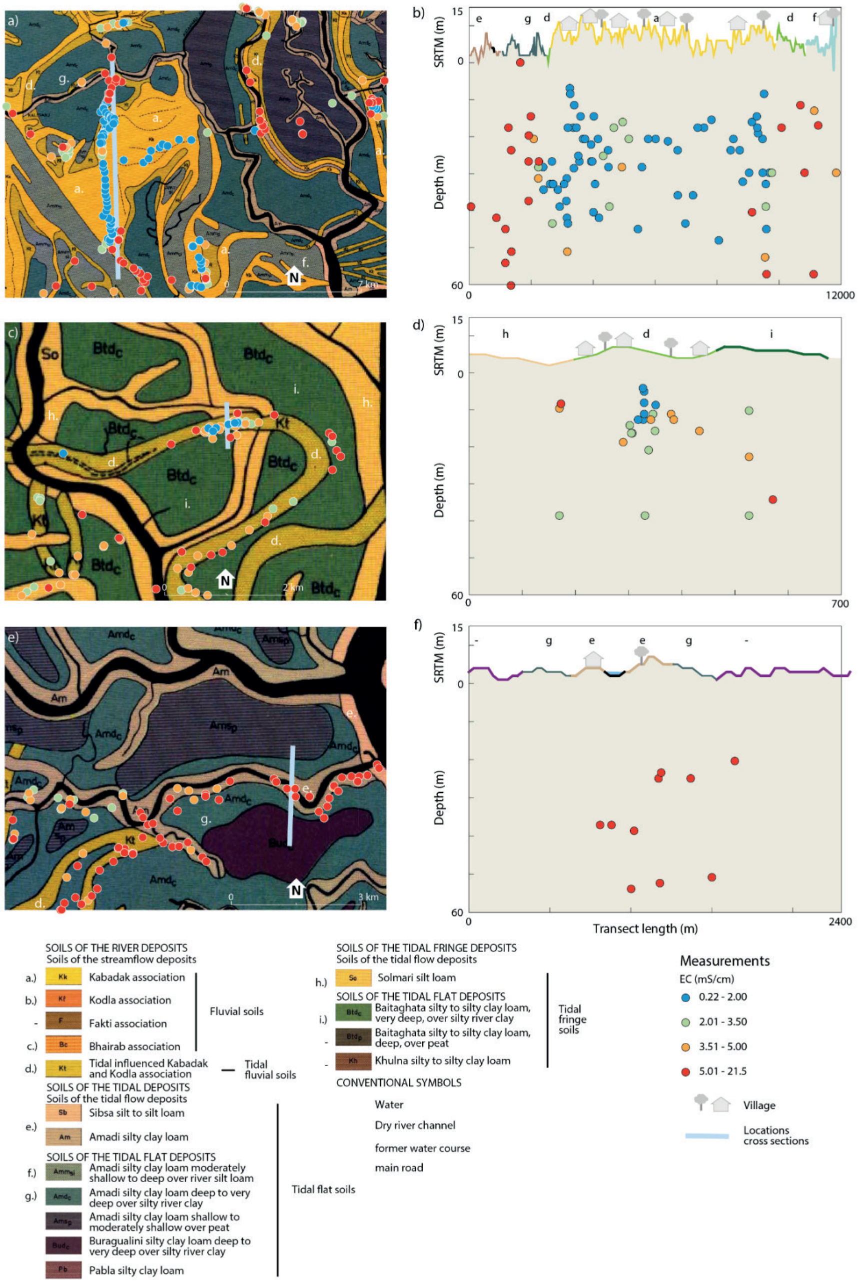


Figure 3.6. Examples of soil landscape features (left) with associated cross section (right). The wide fluvial soils are shown in panels (a) and (b), the tidal fluvial soils are shown panels (c) and (d), and the tidal flat and fringe soils are shown in panels (e) and (f). The underlying map at the left is the soil map from the Food and Agriculture Organization of the United Nations (1959), of which the legend is presented below the figure.

Brackish or saline groundwater was also detected under the fluvial soils. Looking at the soil map, it seems that the areas of fluvial soils with brackish to saline groundwater are narrower than the areas of fluvial soils with fresh groundwater (Figure 3.6c, d). The narrowness could hamper the formation of fresh groundwater lenses, possibly because they are more susceptible to lateral saline groundwater flow or to saline water recharge from occasional flooding. Additionally, some of the fluvial soils with brackish or saline groundwater are adjacent to tidal channels (Figure 3.6c, d), so they might be influenced by flooding or infiltration of saline water from the tidal channel. This could also explain the more saline groundwater in the Bhairab fluvial soil, which occurs next to the tidal Rupsa river (Figure A3.3). Nevertheless, a limited amount of shallow, fresh groundwater was also detected under the narrow strips of fluvial soils.

3.5.2.2 Tidal fluvial soils

The tidal fluvial soils are in low-lying areas surrounding the fluvial soils, or occur as high-lying narrow fossil meanders (Figure A3.3, Figure 3.6e, f). This group sometimes contains shallow fresh groundwater, but only when the soils are at SRTM elevations above approximately 8 m (Figure 3.5a). There are several possible explanations for the tidal fluvial soils having a higher salinity than the fluvial soils. Firstly, as the name suggests, these tidal fluvial soils are formed by tidally influenced rivers, which were probably subjected to some marine influence during sediment deposition, similar to the salinity in the present-day tidal rivers (Bhuiyan and Dutta, 2012). Secondly, they are found in similarly narrow strips as the narrow fluvial soils, suggesting more sensitivity to lateral salinization following saline water recharge in the surrounding low-lying areas (Naus et al., 2019; Paul and Vogl, 2011; Rahman et al., 2018; Sarker et al., 2018). Lastly, it is possible that the relatively small tidal fluvial features were deposited more recently than the large-scale paleo channels, so have had less time to be recharged by fresh groundwater. Nevertheless, some fresh groundwater is present in the shallow part of the aquifer under these tidal fluvial soils on high-lying areas. This is likely the result of recharge with fresh water.

3.5.2.3 Tidal flat and tidal fringe soils

The various tidal flat or fringe soils are usually present as low-lying flats or as very narrow ridges next to contemporary tidal creeks, tidal rivers or roads (Figure A3.3, Figure 3.6e, f). For the soil classes tidal fringe and tidal flat (flow and Amd), there is a lack of correlation between EC and SRTM elevation (Table 3.2). This suggests that the high areas have a similar salinity to the low-lying areas and that freshwater recharge is completely absent in the higher-lying areas of these soils. Both the tidal flat on fluvial soil and tidal fringe flow soil have a negative correlation between EC and SRTM elevation, but the groundwater remains saline in the high areas. The correlation with SRTM elevation, therefore, does not necessarily indicate freshwater recharge in high areas, but instead indicates possible saline water recharge in the low-lying areas (Naus et al., 2019; Paul and Vogl, 2011; Rahman et al., 2018; Sarker et al., 2018). The narrow ridges are expected to be natural tidal creek levees or manmade embankments and roads, both of which would result in the occurrence of mostly clayey material towards the top of the soil profile (Weinman et al., 2008). The local lithology could hamper freshwater recharge in the higher parts of the various tidal flat or fringe soils. Additionally, both the natural levees of the tidal creeks and the embankments have formed relatively recently, which also limits how much influence freshwater recharge might have had. In the tidal flat or fringe soils, what gives guidance for predicting groundwater salinity is not the SRTM elevation but regional trends (Table 3.2). The regional gradient of groundwater increasing in salinity eastwards is similar to the direction the coastline has accreted during the Holocene (Shamsudduha and Uddin, 2007) and could also explain why the tidal fringe soils are more saline than the tidal flat soils. The salinity in the tidal flat soils also correlates with the thickness of the clay layer and with depth, which suggests that possible saline water recharge from the surface mostly affects the part of the groundwater that is less isolated, as we will discuss in Section 3.5.3.

3.5.3 Deeper connate groundwater not affected by landscape features

The inconclusive correlations and large variation in salinity with landscape feature data is expected to be related to the prominent occurrence of connate groundwater (Worland et al., 2015; Ayers et al., 2016; Naus et al., 2019). There are several indications that relatively deep groundwater is less controlled by landscape features than relatively shallow groundwater. Firstly, the EC of deeper groundwater correlates less with SRTM elevation than the EC of shallow groundwater (Table 3.3). Secondly, the shallow groundwater is fresher than the deep groundwater under the fluvial Kobadak and tidal fluvial soils, whereas the shallow groundwater is more saline than the deep groundwater under the tidal flat Amd and tidal flat flow soils, under one-season rice and under aquaculture (Table 3.3). These differences suggest that the deep groundwater could mostly consist of connate water, and that its salinity varies independently from landscape features and is controlled by the hydrological conditions prevailing when the aquifer became sealed off. In Bangladesh, the salinity of the deeper, paleo-controlled groundwater correlates generally more strongly with the regional north-south and west-east gradients than does the salinity of the shallow groundwater. As stated before, these gradients are linked to the distance to the coastline and with the northwest to southeast direction in which the coastline accreted during the Holocene (Shamsudduha and Uddin, 2007). As a consequence, the deep groundwater in tidal flat flow soils in the northwest is fresh. Additionally, in the southern part of the Khulna district, there are some fresh groundwater pockets deeper than 30 m which probably formed under slightly fresher conditions than the surrounding more saline groundwater. To understand and predict the occurrence and the salinity of connate groundwater, it is necessary to have detailed lithological data and detailed understanding of the paleo conditions (Naus et al., 2019). The importance of paleo conditions to understand contemporary groundwater salinity in deeper groundwater has also been reported elsewhere, for example in the Mekong delta (Tran et al., 2012), in Suriname (Groen et al., 2000), and in the Netherlands (Delsman et al., 2014).

3.5.4 Practical guidelines for predicting groundwater salinity in southwestern Bangladesh

The geomorphological analysis above must be seen as a first step towards presenting possible explanations for the regional groundwater salinity variation in southwestern Bangladesh. Even though there remain, uncertainties related to the paleo and present-day hydrological processes, caused by insufficient knowledge of the lithology, recharge rates and groundwater flow, and the occurrence and scale of tidal and marine intrusions, we feel our analysis of relations between groundwater EC and landscape features and the associated controlling hydrological processes provides valuable information for practical applications related to groundwater exploration and for future geohydrological studies in the study area. Based on the above interpretation of the dominant hydrological processes for various landscape features, we therefore constructed guidelines for the prediction of groundwater salinity throughout the region, in the form of a flow chart (Figure 3.7). Soil type, lateral extent of the soil type, elevation, depth below surface and distance to tidal rivers were used as distinguishing factors. The criterion for the width of the areas with fluvial soils was set at approximately 500 m, with the final division being done by expert judgement following visual inspection of the landscape features. The values for SRTM and depth used in the flowchart were estimated based on the patterns revealed by the LOESS results (Figure 3.5).

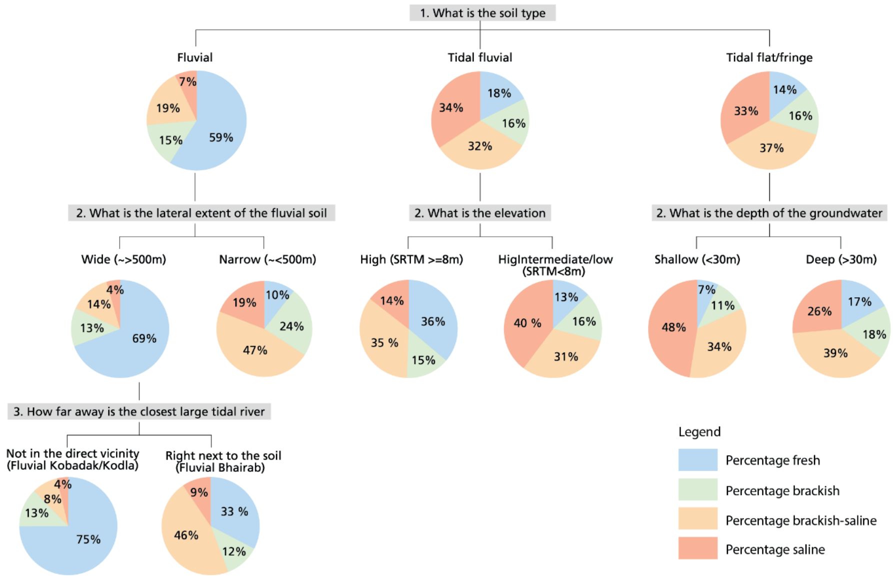


Figure 3.7. Flowchart to identify landscape features to predict the groundwater salinity. Starting from the top it shows leading questions and subsequent answers. The pie charts show the percentages of fresh, brackish, brackish-saline, and saline groundwater measurements for each sub step.

For each distinguished landscape features group, percentages were calculated for the occurrence of the 4 EC salinity classes distinguished. These percentages can be interpreted as the chance of finding each of the salinity classes in a landscape features group. Figure 3.7 illustrates that the best chance of finding fresh groundwater is under fluvial soils (59%), especially under wide areas containing fluvial soils (69%) and far away from tidal rivers (75%). In the rest of the landscape features groups, the chances of finding fresh groundwater are lower and depend on local conditions. In tidal fluvial soils, some fresh groundwater occurs in the high-lying parts, and in tidal flat areas, there is a slight chance of finding connate fresh groundwater whose location cannot be determined based on landscape features. The chance of finding saline water is highest under tidal flat or tidal fringe soils in wells < 30 m deep (48%).

The occurrence of the wide fluvial soils could explain some of the regional groundwater salinity patterns. In the Khulna region, hardly any fluvial soils are present, except for the fluvial soils with some limited fresh groundwater that occur next to the Rupsa river. This lack of fluvial soils reflects the anomaly of more saline groundwater towards the north in Khulna district, and suggests that the reason for this increase in salinity is that for most of the Holocene, Khulna district has consistently been a low-lying area. The lack of paleo channels has also been suggested by Hoque et al. (2014), who described the Khulna region as having been part of the paleo interfluve during the Pleistocene, in contrast to Satkhira and Bagerhat, which were interpreted to be part of the paleo channel areas.

The flow chart uses the fluvial soils with a large lateral extent as a landscape feature factor that indicates high chances for fresh groundwater. The FAO soil map used in this study, however, only covers the western and central parts of the study area (Figure A3.3). The SRTM-elevation map shows areas with a high elevation in the northeastern part of the study area (Figure A3.1), which resemble those of the fluvial fossil meanders with a large lateral extent in the northwest. This suggests similar paleo channel structures in the northeast with high chances for fresh groundwater. Unfortunately, we have too few EC measurements in the northeast to be able to verify this, but our approach illustrates how landscape features in areas without EC measurements can be used to estimate groundwater salinity. The main lines of the approach used here for southwestern Bangladesh may be applied similarly in other coastal areas with available spatial data on elevation, soil types and other landscape feature data, to infer first-order prediction of possible groundwater salinity variation.

3.6 CONCLUSIONS

This study is the first to illustrate the relation between landscape features, hydrological processes and the shallow (<60 m) groundwater salinity throughout southwestern Bangladesh. This knowledge is directly relevant for assessing and overcoming water supply problems in the region. Additionally, the main lines of our approach may be applicable to predict groundwater salinity variation in other coastal areas with available spatial landscape feature data.

We conclude that geomorphologically analysed landscape features can be used to determine which controlling hydrological processes can be expected and to make a first prediction of the groundwater salinity. The chance of fresh water recharge and subsequent occurrence of shallow fresh groundwater is highest (75%) in wide occurrences of fluvial soils type Kodabak and Kodla that are not in the vicinity of tidal rivers. These wide occurrences of fluvial soils are interpreted to be remnants of sandy deposits in large paleo channels. High-lying areas with other soil types are interpreted to be more susceptible to lateral saline water flow or saline water recharge by occasional tidal flooding, and consequently have a lower chance of finding fresh groundwater. The chance of saline water recharge and subsequent occurrence of saline water is highest (48%) in wells < 30 m deep under tidal flat or tidal fringe soils.

Acknowledgements

This work is part of the Delta-MAR project funded by the Urbanizing Deltas of the World programme of NWO-WOTRO (Grant number: OND1357179). We would like to thank Chelsea Peters, Steven Goodbred, John Ayers from Vanderbilt University for allowing us to use their Electrical Conductivity measurements throughout the region. Additionally, we would like to acknowledge the help of all the staff from the Delta-MAR office in Khulna for their support during the fieldwork campaigns, particularly Abir Delwaruzzaman, who always was ready to assist during the fieldwork and who collected some of the EC measurements throughout the region. We would like to acknowledge all MSc students that assisted during the fieldwork, particularly Ruth Wijland, Luc Essink, Kennard Burer, Jan Dirk Smidt from Utrecht University and Atikul Islam and Arif Hasan from Dhaka University. Lastly, Dr Joy Burrough is acknowledged for editing a near-final version of the manuscript.

APPENDIX A

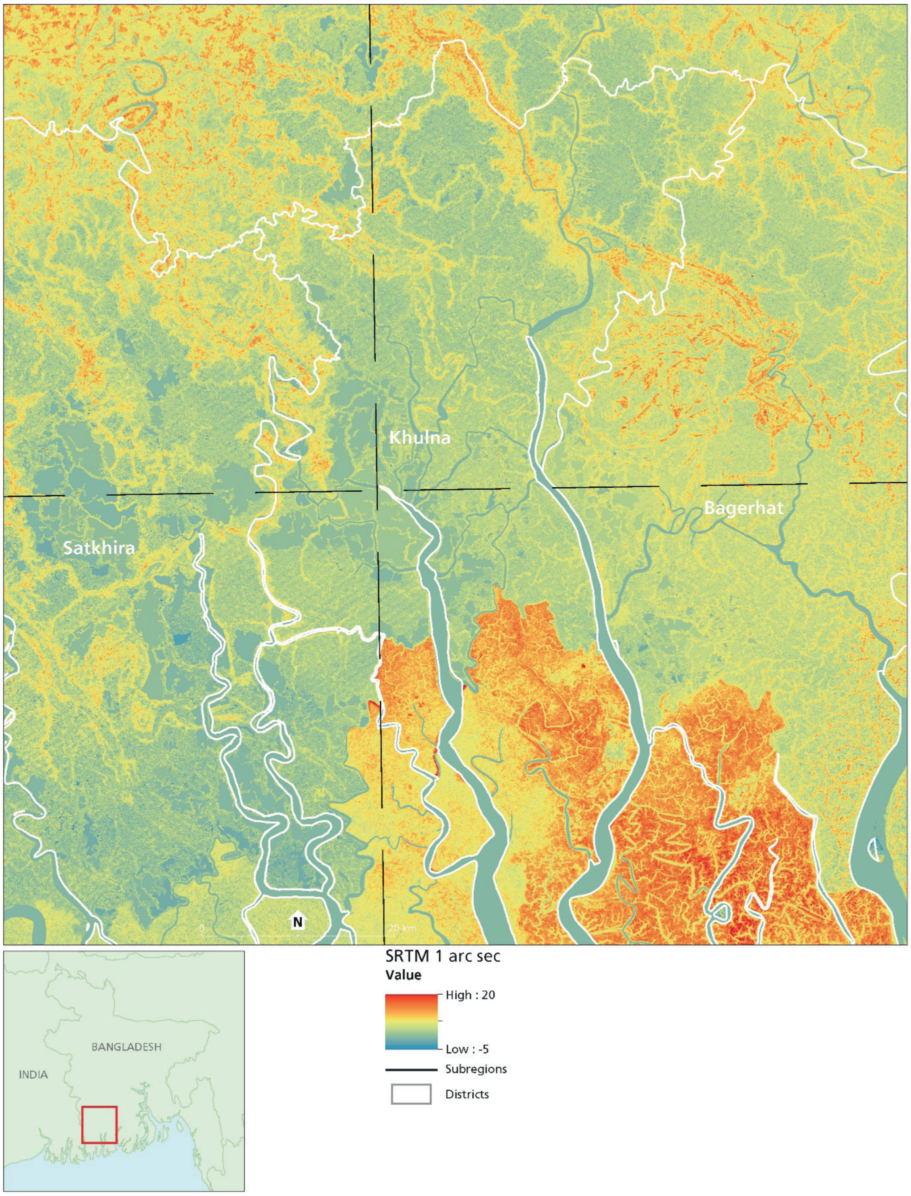


Figure A3.1. Shuttle Radar Topography Mission data (SRTM) in the region (Farr et al., 2007)

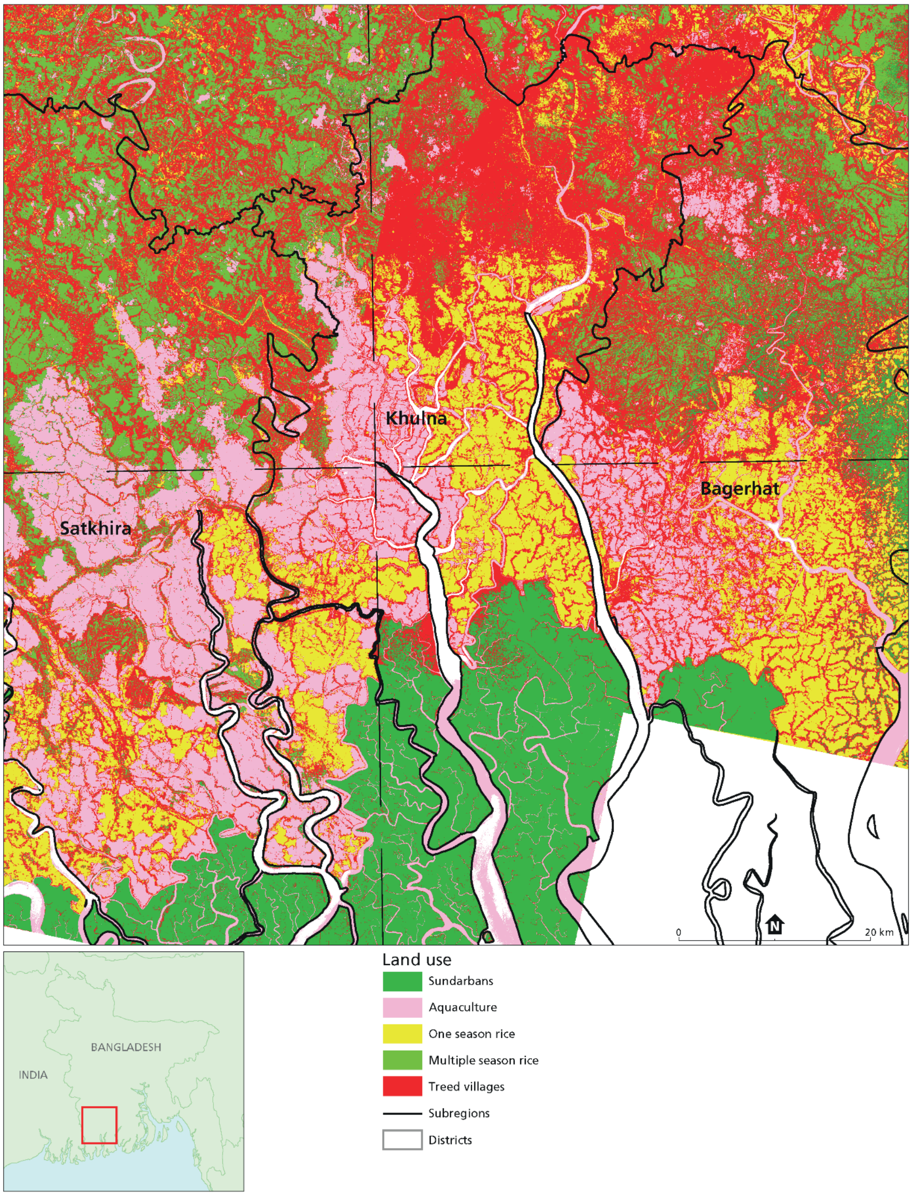


Figure A3.2. Land use in the region based on supervised classification of cloudless Imagery from Landsat 8 (17th and 24th of March 2015, path 137 and 138, row 44), calibrated with observations from aerial pictures and field observations.

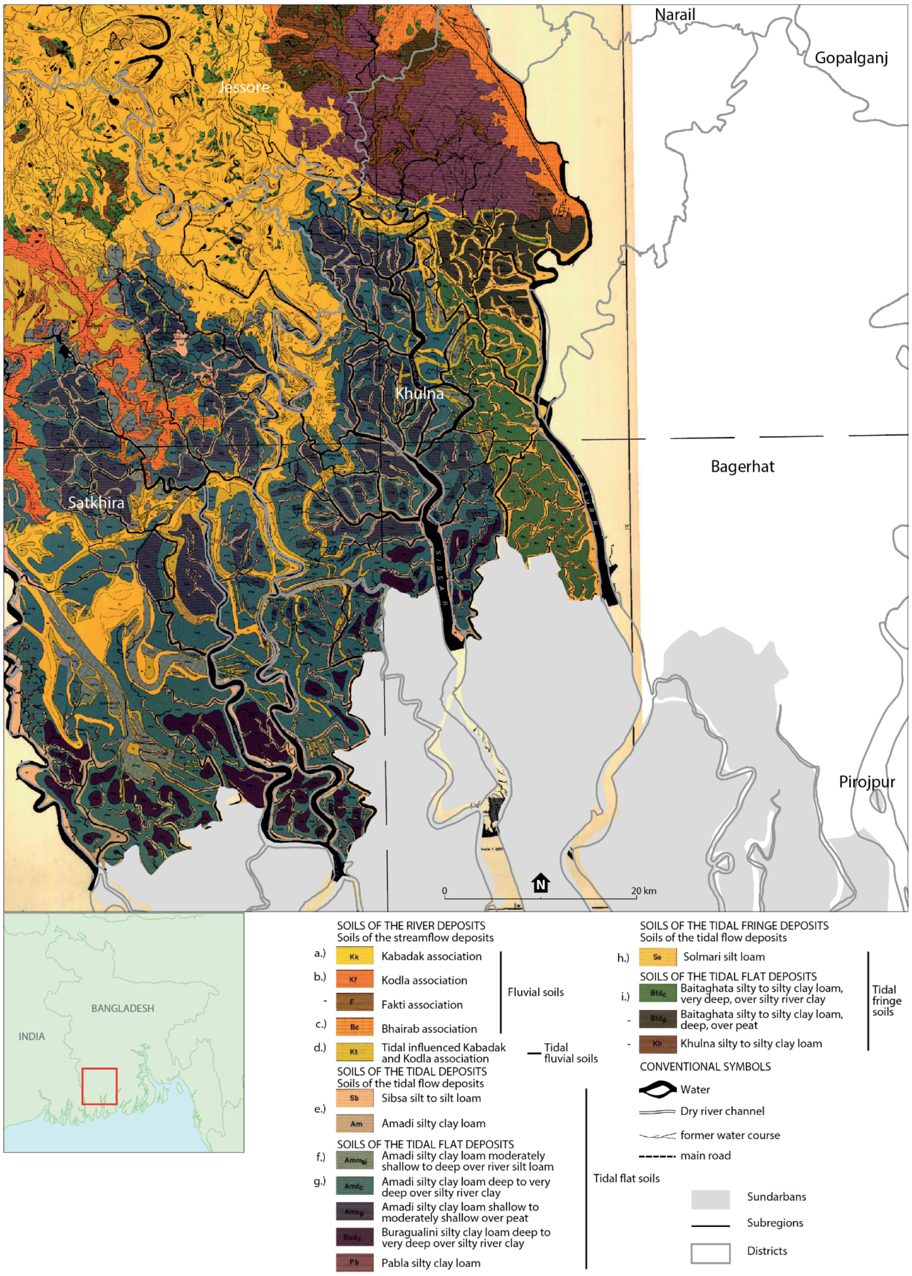


Figure A3.3. Soil map of part of the study area (Food and Agriculture Organization of the United Nations, 1959).

REFERENCES

Alam, M., Sultana, M., Nair, G.B., Sack, R.B., Sack, D. a, Siddique, a K., Ali, A., Huq, A., Colwell, R.R., 2006. Toxigenic Vibrio cholerae in the Aquatic Environment of Mathbaria, Bangladesh. Appl. Environ. Microbiol. 72, 2849–2855. https://doi.org/10.1128/AEM.72.4.2849-2855.2006

Auerbach, L.W., Goodbred Jr, S.L., Mondal, D.R., Wilson, C. a., Ahmed, K.R., Roy, K., Steckler, M.S., Small, C., Gilligan, J.M., Ackerly, B. a., 2015. Flood risk of natural and embanked landscapes on the Ganges–Brahmaputra tidal delta plain. Nat. Clim. Chang. 5, 153–157. https://doi.org/10.1038/nclimate2472

Ayers, J.C., Goodbred, S., George, G., Fry, D., Benneyworth, L., Hornberger, G., Roy, K., Karim, M.R., Akter, F., 2016. Sources of salinity and arsenic in groundwater in southwest Bangladesh. Geochem. Trans. 17, 4. https://doi.org/10.1186/s12932-016-0036-6

Bangladesh Water Development Board, 2013. Hydrogeological Study and Mathematical Modelling to Identify Sites for Installation of Observation Well Nests, Selection of Model Boundary, Supervision of Pumping Test, Slug Test, Assessment of Different Hydrogeological Parameters Collection and Conduct Chemical Analysis of Surface Water and Groundwater, Bangladesh Water Development Board, Dhaka.

Bhuiyan, M.J.A.N., Dutta, D., 2012. Assessing impacts of sea level rise on river salinity in the Gorai river network, Bangladesh. Estuar. Coast. Shelf Sci. 96, 219–227. https://doi.org/10.1016/j.ecss.2011.11.005

Bhuiyan, M.A.H., Rakib, M.A., Dampare, S.B., Ganyaglo, S., Suzuki, S., 2011. Surface water quality assessment in the central part of Bangladesh using multivariate analysis. KSCE J. Civ. Eng. 15, 995–1003. https://doi.org/10.1007/s12205-011-1079-y

Bilgili, A. V., 2013. Spatial assessment of soil salinity in the Harran Plain using multiple kriging techniques. Environ. Monit. Assess. 185, 777–795. https://doi.org/10.1007/s10661-012-2591-3

British Geological Survey (BGS) and Department of Public Health Engineering (DPHE), 2001. Arsenic contamination of groundwater in Bangladesh; BGS Technical Report WC/00/19. British Geological Survey, Keyworth.

Chowdhury, N.T., 2010. Water management in Bangladesh: an analytical review. Water Policy 12, 32. https://doi.org/10.2166/wp.2009.112

Cleveland, W.S., 1979. Robust Locally Weighted Regression and Smoothing Scatterplots. J. Am. Stat. Assoc. 74, 829–836. https://doi.org/10.1080/01621459.1979.10481038

Creel, L., 2003. Ripple effects: Population and coastal regions, Population Reference Bureau.

Custodio, E., Bruggeman, G.A., 1987. Groundwater problems in coastal areas: a contribution to the International Hydrological Programme. Studies and reports in hydrology, vol. 45, UNESCO, Paris.

de Louw, P.G.B., Eeman, S., Siemon, B., Voortman, B.R., Gunnink, J., van Baaren, E.S., Oude Essink, G.H.P., 2011. Shallow rainwater lenses in deltaic areas with saline seepage. Hydrol. Earth Syst. Sci. 15, 3659–3678. https://doi.org/10.5194/hess-15-3659-2011

Delsman, J.R., Hu-A-Ng, K.R.M., Vos, P.C., De Louw, P.G.B., Oude Essink, G.H.P., Stuyfzand, P.J., Bierkens, M.F.P., 2014. Paleo-modeling of coastal saltwater intrusion during the Holocene: An application to the Netherlands. Hydrol. Earth Syst. Sci. 18, 3891–3905. https://doi.org/10.5194/hess-18-3891-2014

Dunn, O.J., 1961. Multiple Comparisons among Means. J. Am. Stat. Assoc. 56, 52–64. https://doi.org/10.1080/01621459.1961.10482090

Dunn, O.J., 1964. Multiple Comparisons Using Rank Sums. Technometrics 6, 241–252. https://doi.org/10.1080/00401706.1964.10490181

Fakhruddin, S.H.M., Rahman, J., 2014. Coping with coastal risk and vulnerabilities in Bangladesh. Int. J. Disaster Risk Reduct. 12, 112–118. https://doi.org/10.1016/j.ijdrr.2014.12.008

Food and Agriculture Organization of the United Nations, 1959, Unger, J. L., Soil Survey of the Ganges-Kobadak Area, Reproduced with permission

Farr, T.G., Rosen, P.A., Caro, E., Crippen, R., Duren, R., Hensley, S., Kobrick, M., Paller, M., Rodriguez, E., Roth, L., Seal, D., Shaffer, S., Shimada, J., Umland, J., Werner, M., Oskin, M., Burbank, D., Alsdorf, D., 2007. The Shuttle Radar Topography Mission. Rev. Geophys. 45, RG2004. https://doi.org/10.1029/2005RG000183

George, G.J., 2013. Characterization of salinity sources in southwestern Bangladesh evaluated through surface water and groundwater geochemical analyses. Master thesis, Vanderbilt University.

Goes, B.J.M., Oude Essink, G.H.P., Vernes, R.W., Sergi, F., 2009. Estimating the depth of fresh and brackish groundwater in a predominantly saline region using geophysical and hydrological methods, zeeland, the netherlands. Near Surf. Geophys. 7, 401–412. https://doi.org/10.3997/1873-0604.2009048

Groen, J., Velstra, J., Meesters, A.G.C.., 2000. Salinization processes in paleowaters in coastal sediments of Suriname: evidence from **δ**37Cl analysis and diffusion modelling. J. Hydrol. 234, 1–20. https://doi.org/10.1016/S0022-1694(00)00235-3

Harvey, C.F., Swartz, C.H., Badruzzaman, A.B.M., Keon-Blute, N., Yu, W., Ali, M.A., Jay, J., Beckie, R., Niedan, V., Brabander, D., Oates, P.M., Ashfaque, K.N., Islam, S., Hemond, H.F., Ahmed, M.F., 2002. Arsenic Mobility and Groundwater Extraction in Bangladesh. Science (80-.). 298, 1602–1606. https://doi.org/10.1126/science.1076978

Hoque, M.A., McArthur, J.M., Sikdar, P.K., 2014. Sources of low-arsenic groundwater in the Bengal Basin: investigating the influence of the last glacial maximum palaeosol using a 115-km traverse across Bangladesh. Hydrogeol. J. 22, 1535–1547. https://doi.org/10.1007/s10040-014-1139-8

Kruskal, W.H., Wallis, W.A., 1952. Use of Ranks in One-Criterion Variance Analysis. J. Am. Stat. Assoc. 47, 583–621. https://doi.org/10.1080/01621459.1952.10483441

Lobell, D.B., Lesch, S.M., Corwin, D.L., Ulmer, M.G., Anderson, K.A., Potts, D.J., Doolittle, J.A., Matos, M.R., Baltes, M.J., 2010. Regional-scale Assessment of Soil Salinity in the Red River Valley Using Multi-year MODIS EVI and NDVI. J. Environ. Qual. 39, 35. https://doi.org/10.2134/jeq2009.0140

McArthur, J.M., Sikdar, P.K., Hoque, M.A., Ghosal, U., 2012. Waste-water impacts on groundwater: Cl/Br ratios and implications for arsenic pollution of groundwater in the Bengal Basin and Red River Basin, Vietnam. Sci. Total Environ. 437, 390–402. https://doi.org/10.1016/j.scitotenv.2012.07.068

Mukherjee, A., Fryar, A.E., Thomas, W.A., 2009. Geologic, geomorphic and hydrologic framework and evolution of the Bengal basin, India and Bangladesh. J. Asian Earth Sci. 34, 227–244. https://doi.org/10.1016/j.jseaes.2008.05.011

Naus, F. L., Schot, P., Groen, K., Ahmed, K. M., and Griffioen, J., 2019. Groundwater salinity variation in Upazila Assasuni (southwestern Bangladesh), as steered by surface clay layer thickness, relative elevation and present-day land use, Hydrol. Earth Syst. Sci., https://doi.org/10.5194/hess-23-1431-2019

Nicholls, R.J., Cazenave, A., 2010. Sea-Level Rise and Its Impact on Coastal Zones. Science. 328, 1517–1520. https://doi.org/10.1126/science.1185782

Nickson, R., McArthur, J., Burgess, W., Ahmed, K.M., Ravenscroft, P., Rahman, M., 1998. Arsenic poisoning of Bangladesh groundwater. Nature 395, 338–338. https://doi.org/10.1038/26387

Paul, B.G., Vogl, C.R., 2011. Impacts of shrimp farming in Bangladesh: Challenges and alternatives. Ocean Coast. Manag. 54, 201–211. https://doi.org/10.1016/j.ocecoaman.2010.12.001

Rahman, A.K.M.M., Ahmed, K.M., Butler, A.P., Hoque, A.M., 2018. Influence of surface geology and micro-scale land use on the shallow subsurface salinity in deltaic coastal areas: a case from southwest Bangladesh. Environ. Earth Sci. 77, 423. https://doi.org/10.1007/s12665-018-7594-0

Rahman, M.M., Hassan, M.Q., Islam, M.S., Shamsad, S.Z.K.M., 2000. Environmental impact assessment on water quality deterioration caused by the decreased Ganges outflow and saline water intrusion in south-western Bangladesh. Environ. Geol. 40, 31–40. https://doi.org/10.1007/s002540000152

Ranjan, P., Kazama, S., Sawamoto, M., 2006. Effects of climate change on coastal fresh groundwater resources. Glob. Environ. Chang. 16, 388–399. https://doi.org/10.1016/j.gloenvcha.2006.03.006

Santos, I.R., Eyre, B.D., Huettel, M., 2012. The driving forces of porewater and groundwater flow in permeable coastal sediments: A review. Estuar. Coast. Shelf Sci. 98, 1–15. https://doi.org/10.1016/j.ecss.2011.10.024

Sarker, M.M.R., Van Camp, M., Islam, M., Ahmed, N., Walraevens, K., 2018. Hydrochemistry in coastal aquifer of southwest Bangladesh: origin of salinity. Environ. Earth Sci. 77, 1–20. https://doi.org/10.1007/s12665-017-7196-2

Shamsudduha, M., Uddin, A., 2007. Quaternary shoreline shifting and hydrogeologic influence on the distribution of groundwater arsenic in aquifers of the Bengal Basin. J. Asian Earth Sci. 31, 177–194. https://doi.org/10.1016/j.jseaes.2007.07.001

Sharma, B., Amarasinghe, U., Xueliang, C., de Condappa, D., Shah, T., Mukherji, A., Bharati, L., Ambili, G., Qureshi, A., Pant, D., Xenarios, S., Singh, R., Smakhtin, V., 2010. The Indus and the Ganges: River basins under extreme pressure. Water Int. 35, 493–521. https://doi.org/10.1080/02508060.2010.512996

Stuyfzand, P.J., 1993. Hydrochemistry and hydrology of the coastal dune area of the Western Netherlands, Ph.D. thesis. Vrije Universiteit of Amsterdam.

Tran, L.T., Larsen, F., Pham, N.Q., Christiansen, A. V., Tran, N., Vu, H. V., Tran, L. V., Hoang, H. V., Hinsby, K., 2012. Origin and extent of fresh groundwater, salty paleowaters and recent saltwater intrusions in Red River flood plain aquifers, Vietnam. Hydrogeol. J. 20, 1295–1313. https://doi.org/10.1007/s10040-012-0874-y

Vacher, H.L., 1988. Dupuit-Ghyben-Herzberg analysis of strip-island lenses. Geol. Soc. Am. Bull. 100, 580–591. https://doi.org/10.1130/0016-7606(1988)100<0580:DGHAOS>2.3.CO;2

van der Sluijs, P., Steur, G.G.L., Ovaa, I., 1965. De bodem van zeeland: Toelichting bij blad 7 van de bodemkaart van Nederland, schaal 1:200 000. Centrum voor Landbouwpublikaties en Landbouwdocumentatie, Wageningen.

Vlam, A.W., 1942. Historisch-morfologisch onderzoek van eenige Zeeuwsche eilanden, Ph.D. thesis, Utrecht Universiteit, Utrecht.

Walraevens, K., Cardenal-Escarcena, J., Van Camp, M., 2007. Reaction transport modelling of a freshening aquifer (Tertiary Ledo-Paniselian Aquifer, Flanders-Belgium). Appl. Geochemistry 22, 289–305. https://doi.org/10.1016/j.apgeochem.2006.09.006

Weinman, B., Goodbred, S.L., Zheng, Y., Aziz, Z., Steckler, M., van Geen, A., Singhvi, A.K., Nagar, Y.C., 2008. Contributions of floodplain stratigraphy and evolution to the spatial patterns of groundwater arsenic in Araihazar, Bangladesh. Bull. Geol. Soc. Am. 120, 1567–1580. https://doi.org/10.1130/B26209.1

Worland, S.C., Hornberger, G.M., Goodbred, S.L., 2015. Source, transport, and evolution of saline groundwater in a shallow Holocene aquifer on the tidal deltaplain of southwest Bangladesh. Water Resour. Res. 51, 5791–5805. https://doi.org/10.1002/2014WR016262

Zahid, A., Afzal Hossain, A.F.M., Hazrat Ali, M., Islam, K. & Abbassi, S.U., 2018. Monitoring the coastal groundwater of Bangladesh. In ''Groundwater of South Asia'', ed. A. Mukherjee. Springer Nature, Singapore, p. 431-451.


Jeans

CHAPTER 4

WHY DO PEOPLE REMAIN ATTACHED TO UNSAFE DRINKING WATER OPTIONS? QUANTITATIVE EVIDENCE FROM SOUTHWESTERN BANGLADESH

Floris Loys Naus

Kennard Burer

Frank van Laerhoven

Jasper Griffioen

Kazi Matin Ahmed

Paul Schot

2020. Water, 12(2), 342.

ABSTRACT

The acceptance of newly implemented, safe drinking water options is not guaranteed. In the Khulna and Satkhira districts, Bangladesh, pond water is pathogen-contaminated, while groundwater from shallow tubewells may be arsenic- or saline-contaminated. This study aims to determine why, as well as the extent to which, people are expected to remain attached to using these unsafe water options, compared to the following four safer drinking water options: deep tubewells, pond sand filters, vendor water, and rainwater harvesting. Through 262 surveys, this study explores whether five explanatory factors (risk, attitude, norms, reliability, and habit) pose barriers to switching from unsafe to safe drinking water options or whether they could act as facilitators of such a switch. Users' attachment to using pond water is generally low (facilitators: risk and attitude. Barrier: norms). Users are more attached to shallow tubewells (no facilitators. Barriers: reliability and habit). The safe alternatives (deep tubewell, rain water harvesting, pond sand filter, and vendor water) score significantly better than pond water and are estimated to have the potential to be adopted by pond water users. Deep tubewell, rain water harvesting, and pond sand filter also score better than shallow tubewells and could also have the potential to replace them. These findings may be used to optimise implementation strategies for safer drinking water alternatives.

4.1 INTRODUCTION

In many cases around the world, it has been found that the acceptance of safe drinking water options varies and is not necessarily guaranteed (Hurlimann and Dolnicar, 2016; Du Preez et al., 2010; Mäusezahl et al., 2009). To achieve the widespread adoption of safer, alternative drinking water options, the importance of so-called 'software activities' to actively promote a behavioural change among users of drinking water systems has been pointed out (Peal et al., 2010). It is contended that when designing a strategy for providing alternative safe drinking water options, it is important to know what keeps people attached to their unsafe drinking water option and what could facilitate a switch from it.

In the Ganges–Brahmaputra–Meghna delta, with a population of over 170 million, unsafe drinking water is used due to drinking water resources being severely stressed. Surface water resources are polluted (Alam et al., 2006; Bhuiyan et al., 2011), meteorological water resources are subject to distinct seasonality (Chowdhury, 2010; Sharma et al., 2010), and shallow groundwater is often contaminated with arsenic (Ayers et al., 2016; Nickson et al., 1998; Harvey et al., 2002; BGS and DPHE, 2001; Gaus et al., 2003). In southwestern Bangladesh, salinity in surface water and groundwater (Ayers et al., 2016; Naus et al., 2019a; Naus et al., 2019b; Worland et al., 2015) puts further pressure on the available drinking water options, leading to the consumption of bacterially contaminated pond water (Knappett et al., 2011) and of shallow groundwater with elevated levels of arsenic and salinity (Flanagan et al., 2012; Khan et al., 2014).

To overcome drinking water quality problems in southwestern Bangladesh, technical solutions have been introduced, often in a supply-driven manner. An example of a technical solution that has been piloted recently, albeit on a limited scale, is Managed Aquifer Recharge (MAR) (Sultana et al., 2014). Drinking water options that are considered relatively safe are scarce; they vary either spatially (i.e., deep tube wells (DTWs), pond sand filters (PSFs), and vendor water), or temporally (i.e., rainwater harvesting (RWH)). As a consequence, the users of unsafe ponds or shallow tube wells (STWs) do not always have safe alternatives. It should be noted that these alternative drinking water options are not always completely safe either, as DTW water is sometimes brackish or saline, PSFs do not always remove all the coliform bacteria (Harun et al., 2012; Kamruzzaman et al., 2006), and the quality of vendor water cannot be guaranteed, as the source of vendor water is not always known (Kjellén and McGranahan, 2006) and the quality of RWH water can deteriorate over time (Despins et al., 2009; Dobrowsky et al., 2014). Generally, though, these alternative options are assumed to be relatively safe, and in this paper the most problematic sources of water in terms of quality are considered to be ponds and STWs.

Similar to many other cases in the world, these safer alternative water options are not always adopted in Bangladesh. It was found that only 36% of arsenic-free drinking water options installed were functional (Kabir and Howard, 2007). Factors contributing to the likelihood of people adopting arsenic-free drinking water options have been extensively researched in Bangladesh (Inauen et al., 2013; Mosler et al., 2010; George et al., 2017). However, so far, no research has focused on factors that cause people to stay attached to unsafe sources such as ponds and STWs. Information on users' attachment to unsafe water options makes it possible to perform ex-ante assessments of whether technical solutions will be successful, i.e., before costly implementation. This is especially valuable for designing a strategy for introducing technical solutions that will address the usage of pond or STW water in southwestern Bangladesh because safe alternatives are often unavailable here.

We set out to research users' attachment to unsafe drinking water options in the Khulna and Satkhira districts in southwestern Bangladesh, using explanatory factors identified from literature. Our aim was to assess factors that pose barriers to switching from unsafe to safe drinking water options and factors that will act as facilitators to such a switch. As a comparison, we investigated users' attachment to the most frequently available safe drinking water options, namely DTWs, PSFs, RWH, and vendor water. We conclude this paper by discussing the opportunities for, and limitations of, the provision of safe drinking water alternatives to replace the unsafe options.

4.2 VARIABLES EXPLAINING VARIATION IN USER ATTACHMENT TO DRINKING WATER SOURCES

Why do people stay attached to unsafe or less safe drinking water options? To identify relevant explanatory variables that could help to answer this question, we consulted the literature explaining variation in the use of water and sanitation systems in developing countries. This literature unequivocally shows that people's drinking water choices result from their individual evaluation of a variety of factors (Du Preez et al., 2010; Mäusezahl et al., 2009; Peal et al., 2010).

The RANAS model (Mosler, 2012) clusters factors thought to affect people's propensity to adopt a water source—e.g., one that is safe—in five separate blocks: Risk, Attitudinal, Normative, Ability and Self-regulation (RANAS) factors. Risk factors describe people's perceived risk of falling sick from drinking the water from their water source. This is related to perceived vulnerability and the expected health effects associated with their drinking water options. The assumption is that people prefer the options that, in their view, pose a lower health risk. Attitude factors describe how people feel about their drinking water option, e.g., how they perceive the water's palatability, the effort of obtaining the water (i.e., collection time), or the price of the water. Here, the assumption is that people prefer options that provide tastier water, require less effort, and are less costly. Norm factors are related to what is perceived to be approved or disapproved of in their immediate social circle, e.g., whether the people who matter to them use similar or different drinking water options. The assumption here is that people prefer options that are being used by most or all of the households around them. Ability factors describe whether people believe they are able to use a drinking water source, and whether they are confident of continuing to do so. The more this is the case, the more likely it is that people will use this option. Self-regulation factors describe the extent to which users can regulate their own behavior—e.g., to switch to a new drinking water option. People who are more confident in this regard can be expected to switch from an unsafe to a safe drinking water option more readily. It is important to realise that what counts is the perception of people. Whether this perception coincides with reality is of less importance.

So far, the explanatory variables proposed by the RANAS model have been used to explain variation in the adoption of a new source of safe drinking water (Heri and Mosler, 2008; Graf et al., 2008; Huber et al., 2012). However, we adopted the RANAS model as a basis to estimate people's attachment to their current safe or unsafe drinking water option. For this purpose, we had to adjust the model. Since we aimed to research the drinking water option that people already use, we assumed that our respondents were able to use this option. So, we removed ability factors from our adapted version of the RANAS model, replacing them with people's perception of the reliability of their current drinking water option. Additionally, the users in this study did not always have a safe alternative, so there was no clear target behaviour that the users needed to self-regulate. Therefore, instead of assessing self-regulation, we assessed whether people think they use their current option(s) out of habit. Our adapted version of the RANAS model—and the operationalisation thereof—is shown in Table 4.1.

Table 4.1. Operationalisation of the variables explaining the variation in people's attachment to their current drinking water source.

Explanatory Factors	Definition	Interview Questions
Risk		
Vulnerability	Risk of arsenic	How high or low do you think is the risk that you will develop arsenicosis? High risk = 1; Some risk = 2; Neutral = 3; No risk = 4
	Health risk	How healthy do you think your drinking water is? Very unhealthy = 1; Unhealthy = 2; Neutral = 3; Healthy = 4; Very healthy = 5
Attitudes		
Instrumental beliefs	Collection Time	How long does it take in minutes to collect the water from the moment you leave the house until you come back (including walking, queuing, collecting)? Very short (<5 min) = 5; Short (5–9 min) = 4; medium (10–29 min) = 3; Long (30–60 min) = 2; Very long (>60 min) = 1
	Cost	How do you feel about the cost of your water? Expensive = 1; Cheap = 2; Free = 3
Affective beliefs	Palatability	How much do you like or dislike the taste of your drinking water? Strongly dislike = 1; Dislike = 2; Neutral = 3; Like = 4; Strongly like = 5
Norms		
Injunctive norm	Neighbours' opinion	Do your neighbours approve or disapprove of your drinking water source? Strongly disapprove = 1; Disapprove = 2; Neutral = 3; Approve = 4; Strongly approve = 5
Descriptive norm	Regular convention	How many people from your community get water from your drinking water source? Few people/less than 10 = 1; Intermediate amount of people/between 10 and 100 = 2; Many people/more than 100 = 3
Reliability		
	Reliability	Will you be able to get water from your drinking water option in a month's time? Very unsure = 1; Unsure = 2; Neutral = 3; Sure = 4; Very sure = 5
Habit		
	Habit	Do you use your drinking water option out of habit? Very unsure = 1; Unsure = 2; Neutral = 3; Sure = 4; Very sure = 5

4.3 METHODS

4.3.1 Sample Selection

Users' attachment to the two unsafe drinking water options (ponds and STWs) and the four most frequently available safe drinking water options (DTWs, PSFs, RWH, and vendor water) was assessed using surveys during a field campaign in January and February 2018 in multiple rural communities throughout the Khulna and Satkhira districts in southwestern Bangladesh. The data was collected by means of a face-to-face survey from 180 local community members who were using the safe and unsafe drinking water options that we were investigating. We are confident that our assumption that shallow groundwater and pond water are unsafe in our study area is justified because in our study region, the bacterial contamination of ponds is a known risk (Knappett et al., 2011) and the consumption of arsenic- and salinity-contaminated water from STWs has been reported (Flanagan et al., 2012; Khan et al., 2014). We based our choice of research location partly on regional maps of arsenic contamination (BGS and DPHE, 2001; Gaus et al., 2003) and salinity contamination (Naus et al., 2019b). The salinity of the STWs was further confirmed by measuring the Electrical Conductivity (EC) of the water. In addition, we aimed to obtain good coverage of the various different water options throughout the Khulna and Satkhira districts (Figure 4.1).

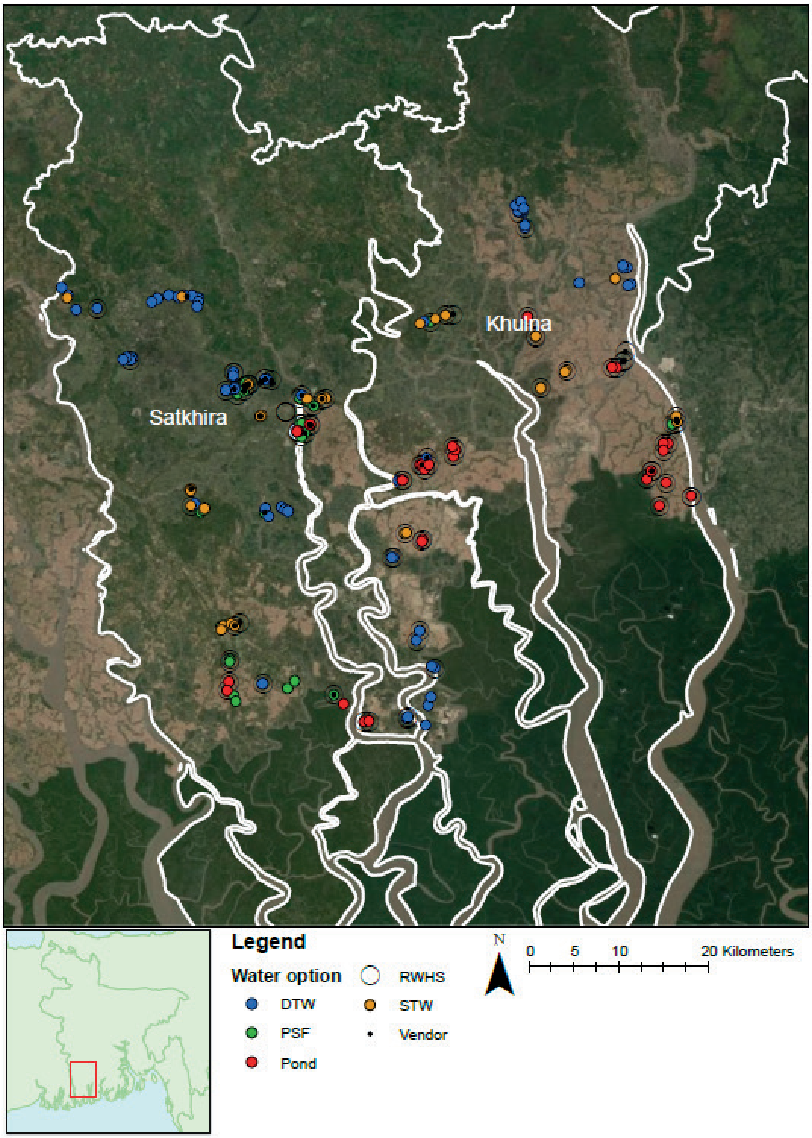


Figure 4.1. Locations of the surveys throughout the region. The symbols and colours indicate the water options related to the survey. DTW = deep tubewell; PSF = pond sand filter; RWHS = rainwater harvesting system; STW = shallow tubewell.

In each community, at least two members were selected for interview, using a random route method. In communities with multiple drinking water options available, more interviews were conducted. Figure 4.1 shows the locations of the interviews. If an interviewee used multiple drinking water options, we aimed to complete a separate questionnaire for each option; this resulted in a total of 262 completed questionnaires. The questionnaire questions were read out in Bengali several times; an interview took between 15 and 40 min, depending on the number of drinking water options the interviewee used. By keeping the length of the questionnaire, on average, around half an hour, we secured the participation of nearly all the households we approached, even though the interviews were unannounced.

4.3.2 OPERATIONALISATION OF THE EXPLANATORY VARIABLES

Both our dependent variable (i.e., drinking water source(s) used) and explanatory variables (i.e., perceived risk, attitude, norms, reliability, and habit) were measured through a structured questionnaire. The full questionnaire can be found in Appendix B. At the start of each interview, before administering the questionnaire, we recorded general information: the number of drinking water options the person used, the GPS location (Figure 4.1), the Electrical Conductivity (EC) of the drinking water option, the monetary costs of the drinking water option, and demographic data on the respondent. Next, the interviewee's perception of the explanatory factors was assessed, using the questions shown in Table 4.1. In Section 4.2, we explained how we identified the factors that might explain the variation in user attachment to drinking water sources. To an important extent, these factors are based on the ground-breaking work by Mosler (2012). For the factor 'risk', we asked about the interviewee's perception of the potability of the water and their perception of the risks associated with arsenicosis. The factor 'attitude' was assessed by the perception of cost, collection time, and palatability. To assess the factor 'norms', we asked what the interviewee thought the neighbours thought about the interviewee's drinking water option, and we asked the interviewee to estimate the size of the community using that drinking water option. To gauge the perceived reliability of the drinking water option, we asked the interviewees whether they thought they would be able to use it in a month's time. Since we administered the questionnaires halfway through the dry season, 'a month's time' corresponded to later in the dry season, when we expected drinking water options would be more limited. Lastly, we asked whether interviewees thought they were using their current water source purely out of habit. For the measurement of most variables, we used a conventional Likert scale rating (with five options, including a neutral option), rather than a forced-choice (ipsative) format. The variables that are less explicitly asking for an opinion (e.g., costs and regular convention), are measured by means of a three-point scale. It has been established that the stability, predictive validity, and concurrent validity of cumulative scores from Likert-type items are independent of the number of scale points utilised (Matell et al., 1971). Therefore, we are confident that the use of various scales has no consequence for the outcome of our analysis.

4.3.3 DATA ANALYSIS

The raw survey data are available as supplementary material. Our analysis is based on the state-of-the-art procedures employed in previous applications of the RANAS model. RANAS has so far only been used to explain the variation in the adoption of new, safe water options. However, we use it to estimate people's attachment to their currently used water options. We therefore developed the following adjusted approach for our analysis.

To be able to compare the explanatory factors, the answers to the questions were normalised, with the lowest or worst perception assigned a value of 1, and the highest or best perception assigned a value of 5. Next, the scores for the questions related to each of the explanatory factors were averaged. The mean and the standard deviation of each answer within each of the drinking water options were calculated. The mean values of the answers reveal the degree of attachment that that particular explanatory factor causes, with higher mean values indicating great attachment and lower mean values indicating little attachment. For the unsafe drinking water options, values with a mean above 4 were seen as possible barriers to change, while values with a mean lower than 3 were seen as possible facilitators of change. For the safe drinking water options, values with a mean higher than 4 were seen as possible opportunities for a switch to the safe drinking water options, while values with a mean below 3 were seen as hampering a switch to the safe drinking water options. The standard deviations indicate the range in the scores between the interviewees.

Because the questionnaires only investigated interviewees' current drinking water options, qualitative interpretation was required to determine the likelihood that interviewees would switch from unsafe to safer drinking water options. To assist in this interpretation, we tested whether the differences in explanatory factors between the drinking water options were significant.

Significant differences between the factors of the different classes were tested using the Kruskal–Wallis test (Kruskal and Wallis, 1952) and Dunn's test (Dunn, 1964).

4.4 RESULTS

4.4.1 General Data

Table 4.2 shows general data obtained by the survey. It can be seen that rather than committing exclusively to one drinking water option, households in Bangladesh often use a portfolio of sources that, in varying ways and to varying extents, satisfy one or more of the several preferences they have with regard to their drinking water (Hasan et al., 2019). For pond and STW water (the two unsafe drinking water options), there were, respectively, 30 and 33 completed questionnaires. The numbers of surveys completed for the alternative safe drinking water options, PSF and vendor water, were similar: 34 and 30, respectively. The sources for which there were the most completed questionnaires were DTWs (70) and RWH (66). DTW is most often the sole drinking water option (69%), followed by STW (44%) and PSF (35%). Almost never used as the sole drinking water option are RWH (1%), pond water (7%), and vendor water (7%).

Table 4.2. General data for each drinking water option (1 USD = 84.5 BDT on 4 July 2019). STW = shallow tubewell, DTW = deep tubewell, RWH = Rainwater harvesting.

Option	No. of Completed Questionnaires	Percentage Sole Drinking Water Option	Multiple Drinking Water Options	Median Cost (BDT/L)	Average Cost (BDT/L)	Usage L/(Day, Person)
Pond	30	7%	93%	0.006	0.335	3.1
STW	32	44%	56%	0	0.031	4.3
DTW	70	69%	31%	0	0.077	3.1
RWH	66	1%	99%	0	0	2.6
PSF	34	35%	65%	0.022	0.131	3.1
Vendor	30	7%	93%	0.833	0.975	2.5

The average amount of drinking water used is 3.2 L per person per day, with only slight differences between the drinking water options: amounts of vendor and rainwater are slightly less, and amount of STW water is slightly more (Table 4.2). These amounts are comparable to those reported in a previous study (Islam et al., 2013), which found the median amount of consumed water to be 3.35 L per person per day. The total water use, including water for cooking and cleaning, is surely larger than these reported amounts, as often there is a separate water option for these other household activities.

The costs vary between the drinking water options. DTW, RWH, and STW are cheap, with a median of 0 BDT per litre (BDT/L) (1 USD = 84.5 BDT on 4 July 2019) and an average below 0.1 BDT/L. Pond and PSF are slightly more expensive: on average between 0.1 BDT/L and 0.35 BDT/L. The costs of using pond water are for treatment using alum as a disinfectant (Sirajul Islam et al., 2007). Vendor water is the most expensive, with an average close to 1 BDT/L (Table 4.2). The responsibility for collecting water is most often shared by all members of the family (45%), followed by being a responsibility for women only (25% of cases), and being a responsibility for men only (14% of cases). In the remaining cases, water was delivered to the home (10%) or collected by servants (4%). Our finding that the responsibility for water collection is most often shared by the members of the household contrasts with the common finding that water collection is mostly a task for the female population (Faisal and Kabir, 2005).

4.4.2. Explanatory Factors

The means and standard deviations of the explanatory factors are presented in Table 4.3 for each drinking water option. The scores for the separate questions are shown in Table A4.1, in Appendix A. In general, only a few facilitators were identified: they scored below the neutral value of 3. Barriers (a score higher than 4) are more common, with each drinking water option having at least one. The standard deviations are often relatively large, which indicates that a neutral value close to 3 will still be judged negatively by an appreciable number of people.

Table 4.3. Standardised mean values of the explanatory factors. For the unsafe drinking water options, the facilitators (values lower than 3) are highlighted in blue, while the barriers (values higher than 4) are highlighted in yellow. For the safe drinking water options, barriers (values lower than 3) are highlighted in orange, while opportunities (values higher than 4) are highlighted in teal. As a reminder, the operationalisation of the factors is summarised below the table.

	Unsafe Drinking Water Options				Safe Drinking Water Options							
Explanatory Factor	Pond, n = 30		STW, n = 32		PSF, n = 34		Vendor, n = 30		DTW, n = 70		RWH, n = 66	
	Mean	STD	Mean	STD	Mean	STD	Mean	STD	Mean	STD	Mean	STD
Risk =	2.28	1.06	3.28	0.98	3.69	0.95	3.87	0.69	3.89	0.95	4.47	0.63
Attitude &	2.59	0.66	3.81	0.71	3.21	0.82	2.86	0.73	3.95	0.76	4.81	0.33
Norms ^	4.22	0.68	3.5	1.09	4.49	0.29	4.21	0.54	4.06	0.73	3.81	0.74
Reliability #	3.73	1.48	4.16	0.99	3.27	1.35	3.27	1.43	4.02	1.05	2.95	1.32
Habit ~	3.27	1.31	4.23	0.82	3.91	0.97	3.39	1.29	4.3	0.76	4.57	0.83

= Risk of arsenic, Health risk, & Collection time, Cost, Palatability, ^ Neighbours' opinion, Regular convention # Reliability, ~ Habit.

For ponds, we identified one barrier and two facilitators for switching to an alternative to this source of drinking water. The barrier is the factor 'norms', with a value of 4.22, which shows that people drink the pond water because the community generally approves of the option and that many people in the community drink it. Risk and attitude are facilitators of a switch: both score less than 3 (Table 4.3). In more detail, people perceive the pond water to be unhealthy; remarkably, they perceive that they are at risk of arsenic poisoning when drinking pond water. Additionally, people do not like its taste, judge the collection time to be long, and perceive the pond water to be expensive (Table A4.1). The combination of the negative attitude towards the ponds, the negative perception of risk and the low percentage of people that use ponds as their sole drinking water option (Table 4.2) suggests that people drink pond water when other drinking water options become unavailable, and not because they want to drink it.

No facilitators were identified for STWs, and habit and reliability were identified as barriers for a switch away from them. People use the STW water out of habit and because it is reliably available throughout the year. The two barriers and the lack of facilitators indicate that it is generally difficult to get

STW users to switch away from their water source. Risk is the least important explanatory factor, with a mean somewhat above the neutral value of 3, indicating that emphasising the risks associated with STW water might be a way to get people to switch to a safer source of drinking water. Compared to ponds, the only factor that is lower for STWs is norms: the probable reason is that most people usually have their own private STW for their exclusive use. When the STWs were grouped according to their salinity, slight differences in the explanatory factors were observed (Table 4.4). The brackish STWs (>2 mS/cm) have a higher score for reliability than the fresh STWs (<2 mS/cm), while fresh STWs have a higher attitude score than the brackish STWs (Table 4.4).

Table 4.4. Differences in explanatory factors between brackish and fresh shallow tubewells. Barriers (values higher than 4) are highlighted in yellow. The colours and symbols in the last column indicate significant differences and which group is higher: Brackish STW (teal +) or fresh STW (purple *). As a reminder, the operationalisation of the factors is summarised below the table.

Explanatory Factor	Brackish STW	n = 13	Fresh STW	n = 15	Significance
	Mean	Std	Mean	Std	Dunn's Test
Risk =	3.19	1.03	3.34	0.97	
Attitude &	3.35	0.70	4.12	0.54	*
Norms ^	3.62	0.87	3.42	1.23	
Reliability #	4.62	0.87	3.84	0.96	+
Habit ~	4.42	1.00	4.11	0.68	

= Risk of arsenic, Health risk, & Collection time, Cost, Palatability, ^ Neighbours' opinion, Regular convention # Reliability, ~ Habit.

For PSFs, scores for 'norms' are higher than 4, indicating that, similar to the ponds, PSFs are approved by the community and many people in the community drink water from them. The PSFs score the lowest for attitude and reliability, which score only slightly higher than the neutral score of 3. Examining the score for attitude in more detail reveals that it is controlled by a negative score for collection time and cost (Table A4.1). For vendor water, attitude scores lower than 3 and norm scores are above 4. The negative attitude is mostly attributable to the costs of the vendor water (Table A4.1), which suggests a preference for using cheaper unsafe drinking water options when they are available. The explanatory factors reliability and habit are only somewhat higher than the neutral score of 3, which—together with the low percentage of people solely using vendor water (Table 4.3)—suggests that the use of vendor water is incidental, despite the high value of norms. For DTWs, almost all factors score higher than 4. Habit, norms, and reliability score just above 4, but attitude and risk score just below 4. These positive scores for all explanatory factors reflect the fact that DTW is often the sole drinking water option used (Table 4.2). For RWH, risk, attitude, and habit are higher than 4, while norms scores just under 4. Reliability scores negatively, below 3. This can be attributed to the large seasonal fluctuations in the availability of rainwater, which result in almost all the people who use rainwater being unable to use it year-round (Table 4.2).

4.4.3. Significant Differences Found for Explanatory Factors

The results of the Dunn's tests are presented in Table 4.5. The unsafe pond water has many factors that score significantly more negatively than the factors of the safe drinking water options. Compared to DTW and RWH, pond has significantly lower values for risk, attitude, and habit, which suggests that there are multiple explanatory factors that could facilitate a switch from pond to DTW or RWH when these options are available. Compared to PSF and vendor, only risk is significantly lower, suggesting that when focusing on the risks of consuming pond water, users will be most amenable to switching from pond water to PSF or vendor water. Pond water does score significantly higher for norms than RWH does, suggesting that a change from pond water to rainwater might encounter resistance from the community. Compared to the other unsafe water source, STW, attitude is significantly more negative for pond water, indicating that users are likely to switch from pond water to STW water because STW water is more convenient to use than pond water.

Table 4.5. Significant differences in explantory factors between the drinking water options. Empty cells indicate non-significant pairs. The colours and symbols indicate, for the significant pairs, which group is higher: the first (teal +) or the second (purple *).

Drinking Water Options	Group1	Group2	Risk =	Attitude &	Norms ^	Reliability #	Habit ~
			p-Value for Each Explanatory Factor				
Unsafe	Pond	STW		*	+		
	Pond	PSF	*				
	Pond	Vendor	*				
	Pond	DTW	*	*			*
	Pond	RWH	*	*	+		*
	STW	PSF			*		
	STW	Vendor		+			
	STW	DTW					
	STW	RWH	*	*		+	
Safe	PSF	Vendor					
	PSF	DTW		*			
	PSF	RWH	*	*	+		*
	Vendor	DTW		*			*
	Vendor	RWH	*	*			*
	DTW	RWH	*	*		+	

[=] Risk of arsenic, Health risk, [&] Collection time, Cost, Palatability, [^] Neighbours' opinion, Regular convention [#] Reliability, [~] Habit.

STW water has significantly lower scores for risk and attitude than RWH and a significantly lower score for norms than PSF. STW users are most likely to use RWH water because RWH is more convenient to use and because of the perceived risk associated with drinking STW water. However, the greater reliability of the STWs compared with RWH does hamper a complete switch from STW water to RWH water. PSF scores significantly higher than STW for the factor 'norms', suggesting that a switch from STW to PSF has the highest chance of occurring when the community aspect of using PSFs is emphasised. Compared to vendor water, STW scores significantly higher for attitude, suggesting that the convenience of STW could limit a possible switch to vendor water. There are no significant differences between STW and DTW water, which indicates that switching from STW to DTW may not be easy, but also that this switch is not necessarily hindered by any of the explanatory factors studied. It may, therefore, be possible to convince the community to switch from STWs to DTWs by 'software' interventions (Peal et al., 2010).

There are also some significant differences between the improved drinking water options. PSF and vendor water score lower than DTW for attitude and lower than RWH for risk, attitude, and habit, suggesting that DTW and RWH would also have social potential to be adopted by people who use vendor or PSF water.

4.5 DISCUSSION

The aim of our paper was to assess the attachment of users to their currently used unsafe water options. This information is valuable for ex-ante assessments of whether technical solutions will be successful and, consequently, for strategies introducing technical solutions that will address the usage of pond or STW water in the Khulna and Satkhira districts in southwestern Bangladesh, especially because safe alternatives are often unavailable here. Here, we will first discuss the implications of our findings for the potential for replacing the unsafe drinking water options. Next, we will discuss the potential and limitations of the most frequently present alternative drinking water options to replace the unsafe drinking water options.

4.5.1 Improvement Strategies

4.5.1.1. Pond

A total of 17% of our respondents indicated that they currently use pond water for drinking water purposes (Table 4.2). The vast majority of pond water users (93%) use it in combination with other sources. A small proportion (7%) report that this is the sole source supplying their drinking water needs (Table 4.2). How can pond water users be pulled away from this relatively unsafe source? Our results suggest that the low scores for risk and attitude (Table 4.3) could be exploited: pond water users are aware of the risks associated with drinking pond water, and it seems possible that they could be persuaded to switch given that the scores for the risks associated with the sources PSF, vendor, DTW, and RWHS are significantly lower (Table 4.5). Additionally, the low risk score suggests that pond users would also be amenable to switching to other safe lower-risk alternatives that we did not investigate. The significantly lower score for the attitude to pond water compared to the scores for the attitude to DTW and RWHS is further evidence that a switch to these options might be achieved (Table 4.5). The overall low score for attitude suggests there is great potential for pond water users to switch to a more palatable, less time-consuming or cheaper alternative drinking water option (Tables 3 and A1). The safe alternative options would not have to be superior in all the inconveniences responsible for the low scores for the attitude to pond water (Table A4.1). For example, the finding that users of pond water often pay for their water suggests that they are able and could be willing to pay for alternative, safe drinking water options too (Tables 2 and A1). The alternative water sources, therefore, do not necessarily have to be free. The same applies to the collection time: the time taken to collect water from ponds is long (Table A4.1), so alternative, safe drinking water options sited centrally could still be attractive to pond water users.

The relatively high score for norms for pond water (Table 4.3) could inhibit the switch away from pond water, especially as previous studies have found that social norms greatly influence which drinking water option is chosen (Mosler et al., 2010; Graf et al., 2008; Huber et al., 2012; Altherr et al., 2008; Moser and Mosler, 2008). When providing alternative, safe drinking water options, a campaign focusing on changing the social norms may be necessary to get people to adopt the new safe options (Peal et al., 2010). In general, the findings that pond water users take a long time to collect their water, do not like the taste of the water, think that it is expensive to pay for the addition of alum (Table A4.1), and have a negative perception of the pond water potability (Table 4.3) suggest that many of them would be very willing and motivated to switch to safer water options, but currently have no option to do so.

4.5.1.2. Shallow Groundwater

A total of 18% of our respondents indicate they currently obtain drinking water from STWs (Table 4.2). A total of 44% of the households using STWs report that this is their sole source of drinking water (Table 4.2). Our findings suggest that the following might help achieve a switch from STW to safer options. Given that the scores for the explanatory factor attitude are significantly higher for STW users than for pond water users and that, compared to pond, STW scores significantly lower less often than the safe water options (Table 4.5), switching to safe drinking water options would probably be more difficult to achieve for STW users than for pond users.

Our findings show that when PSF systems are introduced, the best strategy to achieve adoption is probably to emphasise and change the community norms (Table 4.5), whereas when RWH systems are introduced, the best strategy is probably to emphasise the health risks associated with STW use (Table 4.5). As the risk associated with STWs scores the lowest of all the explanatory factors (Table 4.3), emphasising the risk may also achieve the best adoption result for other safe alternatives not investigated in our study. This contention is supported by the finding that, elsewhere in Bangladesh, highlighting the risk of arsenic caused people to change their water source in 65% of cases (Opar et al., 2007). Unfortunately, in that study, most of the people changed to new, untested STWs, which shows that achieving a switch to completely different alternative drinking water options is not straightforward; it could be hindered by the availability of alternative safe drinking water options. For STW users to change their behaviour, there must be other safe options nearby (George et al., 2017).

Given our finding that the barriers responsible for STW users remaining attached to their water source are reliability and habit (Table 4.3), the alternative safe water source should also be reliably available. The positive score for habit could be the largest barrier for a switch away from using STW water. Habits are hardest to change (Mosler, 2012) and could prevent people switching away from STW water, even if they are willing to switch. In conclusion, the lack of clear facilitators, together with the clear barriers of reliability and norms, suggests that it is difficult to tackle the water quality problems associated with STW water. Lastly, it should be noted that there are differences in scores between brackish STWs and fresh STWs: for fresh STWs, attitude is valued significantly higher, whereas for brackish STWs, reliability is valued significantly higher for brackish STWs (Table 4.4), suggesting that different implementation strategies are required, depending on the salinity of the STW.

4.5.2. Implementation Potential of the Investigated Alternatives

Aside from the potential based on the explanatory factors investigated, other factors might hamper a switch from a currently used unsafe source to alternative safe drinking water options. Here, we discuss the potential of the investigated alternatives to replace the unsafe drinking water options by taking other factors into account.

4.5.2.1. Rainwater Harvesting (RWH)

A total of 37% of our respondents report that they use RWH to satisfy their drinking water needs (Table 4.2). The vast majority (99%) of RWH users use it in combination with other options (Table 4.2). Harvesting rainwater is cheap and easy but storing it for a longer period of time is a challenge. RWH scored significantly better than pond for risk, attitude, and habit and scored significantly higher than STW for risk and attitude (Table 4.5), suggesting there is great potential for RWH to be adopted by both pond and STW users, as was also found by a previous study (Peters et al., 2019). However, it is only available cheaply in the wet season, July to October. Most pond water users, and some of the STW users, already drink rainwater in the wet season. To overcome the seasonality of rainwater availability, larger reservoirs could be constructed. These require a large financial investment, and the stored rainwater will be more susceptible to the deterioration of quality (Despin et al., 2009; Dobrowsky et al., 2014). Nevertheless, rainwater is available throughout the region, so the potential to implement RWH systems is everywhere, provided there is space for the reservoirs. This also suggests that the need for other safe water sources is limited to the dry season. This is important for the provision of the other safe drinking water options. If other alternatives need to rely on the financial contributions of the users, income can only be expected in the dry season. It should be noted that, given that most DTW users do not use rainwater throughout the year (Table 4.2), other water sources could completely replace rainwater, provided they score well enough (Table 4.3).

4.5.2.2. PSFs

A total of 19% of our respondents indicate they use PSF (Table 4.2). Of all PSF users, 35% of the households use it as their only source of drinking water (Table 4.2). Our findings suggest that PSFs are likely to be adopted by pond users if differences in the perceived risks are emphasised (Table 4.5). The PSFs can be constructed next to currently used ponds, which could facilitate the switch from pond to PSFs. However, as already noted, the current design of PSFs is unable to remove all coliform bacteria (Harun and Kabir, 2012; Kamruzzaman et al., 2006), so there is a need for improved filters. A second drawback is that the PSFs require an initial financial investment, as well as additional financial contributions for maintenance. As a consequence of this, many previously installed PSFs have been abandoned. In a study of arsenic mitigation technologies in southeastern Bangladesh (Hossain et al., 2015), the levels of abandonment of PSFs were found to be 87%. Even though the PSFs score significantly better than STWs for norms (Table 4.5), PSFs are less easily adopted as an alternative to arsenic-contaminated STWs. Filter plants were found to be abandoned within weeks of construction (Hoque et al., 2004) and, when attempts were made to replace arsenic-contaminated STWs, only a moderate acceptance of PSFs was found (Inauen et al., 2013).

4.5.2.3. DTWs

As many as 39% of our respondents indicate that they rely on DTWs to cover their drinking water needs (Table 4.2). Of all DTW users, 69% indicate that they use no other source (Table 4.2). The largely positive factors suggest that DTWs have great potential to replace unsafe drinking water options at locations where deep groundwater is of sufficient quality (Table 4.3). The significantly higher scores for risk, attitude, and habit suggest that DTWs have great potential to be adopted by pond users (Table 4.5). We found no factors preventing or facilitating the replacement of STWs by DTWs (Table 4.5). The high potential of DTWs was also found previously (Inauen et al., 2013; Heri and Mosler, 2008; Hoque et al., 2004). The main limitation for DTWs remains the spatial availability of deep fresh groundwater. To some extent, the spatial limitation can be overcome by using piped systems to distribute deep groundwater from locations with deep fresh groundwater throughout the region. These piped systems have been found to be well accepted (Inauen et al., 2013; Hoque et al., 2004) and therefore have potential to be accepted as an alternative to ponds or STWs. However, the initial investment and maintenance costs are even higher than for the construction of manually operated DTWs. Additionally, it should be noted that fresh groundwater recharge is limited (Ayers et al., 2016; Naus et al., 2019a), which means that the use of deep groundwater would probably be unsustainable. Nevertheless, the mining of deep groundwater could be a useful solution for the short term. Before large infrastructure investments are made, the availability and sustainability should be researched in detail.

4.5.2.4. Vendor Water

A total of 17% of our respondents rely on water vendors, but relatively few of them (7%) rely on them solely (Table 4.2). Given the significantly higher score for the factor risk (Table 4.5), vendor water has the potential to replace ponds if the differences in risk between vendor water and pond water are emphasised. The significantly lower score for attitude compared to STWs suggest that it is unlikely that vendor water can replace STWs. The vendor water market remains non-transparent, and throughout the region it is often unclear what the source of the vendor water is (Kjellén and McGrahan. 2006). Additionally, the fact that vendor water is expensive shows that it cannot fully replace the use of unsafe water, as people who cannot afford to pay for it have no choice but to use the unsafe, but cheap or free, drinking water options.

4.6. CONCLUSIONS

We assessed the attachment of users of unsafe water to their current drinking water option. Pond users are not very attached to their drinking water option and therefore it would be relatively easy to get people to switch from drinking pond water to safer alternatives. Compared with pond water users, STW users are more attached to their water source, indicating that it is expected to be more difficult to stop people consuming groundwater contaminated by salinity or arsenic than to stop them consuming bacterially contaminated pond water. This difference in attachment implies that efforts to improve public health may be most effective when focusing on users of pond water.

The greatest chance of getting people to switch from pond water is expected to be to focus on the risk and inconvenience of drinking pond water. The largest potential for getting STW users to adopt alternative, safe options is to focus on the risk associated with drinking groundwater with elevated levels of arsenic or salinity. The safer alternatives DTW, RWH, PSF, and vendor water are all estimated to have some potential to be adopted by pond water users, while RWH, DTW, and PSF could also replace STWs. However, the alternatives require large financial investments to make them available throughout the year and throughout the administrative districts of Khulna and Satkhira in southwest Bangladesh.

This paper illustrates the usefulness of research on why people are attached to their unsafe drinking water options and on why people do not adopt safe alternatives. Insight into users' attachment to unsafe drinking water options could be used for ex-ante assessments of the likelihood of successfully introducing a future technical solution and for determining which factors to focus on. Based on the insights we gained, we suggest that future research seeks to compare respondents' perceptions of the explanatory variables, with objective measurements of the same. Additionally, we suggest combining quantitative analysis with more qualitative interpretation, for example, by means of data collection through semi-structured interviews and open-questions.

Acknowledgments: This work is part of the Delta-MAR project funded by the Urbanising Deltas of the World (UDW) programme of the Dutch Research Council (NWO-WOTRO) (Grant number: OND1357179). We would like to acknowledge the help of all the staff from the Delta-MAR office in Khulna for their support during the fieldwork campaigns, particularly Abir Delwaruzzaman, who helped immensely with facilitating the fieldwork. Additionally, we would like to thank all MSc students that assisted during the fieldwork, particularly Mahfuz, Arif, Bijoy and Refrat from Dhaka University. Joy Burrough is acknowledged for editing a near-final version of the manuscript. Lastly, we would like to thank Stevan Wang as editor and anonymous reviewer 1, 2 and 3 for their time and useful comments that helped improve the manuscript.

Compliance with Ethical Standards: Prior to each interview, the explicit consent of the human participant was sought.

APPENDIX A

Table A4.1. Standardised mean values for the answers to the different questions. For the unsafe drinking water options, the facilitators (values lower than 3) are highlighted in blue, while the barriers (values higher than 4) are highlighted in yellow. For the safe drinking water options, barriers (values lower than 3) are highlighted in orange, while opportunities (values higher than 4) are highlighted in teal.

	Unsafe Drinking Water Options				Safe Drinking Water Options							
Question	Pond, n = 30		STW, n = 32		PSF, n = 34		Vendor, n = 30		DTW, n = 70		RWH, n = 66	
	Mean	STD	Mean	STD	Mean	STD	Mean	STD	Mean	STD	Mean	STD
Arsenic risk	2.88	1.41	3.78	1.25	4.00	1.35	4.12	0.83	3.83	1.22	4.65	0.69
Health	2.03	1.13	3.00	1.02	3.50	0.99	3.66	0.86	3.80	1.04	4.42	0.70
Collection time	2.63	1.22	3.87	1.26	2.91	1.40	3.31	1.62	3.70	1.36	5.00	0.00
Cost	2.48	1.63	3.83	1.47	2.88	1.34	1.36	0.78	4.07	1.44	4.88	0.70
Palatabi-lity	2.63	0.85	3.69	0.90	3.85	0.74	3.80	0.85	4.04	0.86	4.64	0.57
Neigh-bours' opinion	3.77	0.82	3.59	1.04	4.09	0.45	3.86	0.64	4.04	0.75	4.09	0.46
Commu-nity size	4.79	0.83	3.41	1.80	5	0	5	0	4.18	1.16	1.67	1.45
Reliability	3.73	1.48	4.16	0.99	3.27	1.35	3.27	1.43	4.07	1.06	2.95	1.32
Habit	3.27	1.31	4.23	0.82	3.91	0.97	3.39	1.29	4.36	0.73	4.57	0.83

4

APPENDIX B

Questionnaire

General Observations

I. Location of interview: House//Street//Other
II. Date of interview
III. Gender: M//F
IV. Land use: Aquaculture (shrimp)//homestead//Crop culture (rice)
V. Distance to paved road: far away//close by//at road
VI. Wealth: Very Poor//Poor//Medium//Rich//Very Rich
VII. GPS location:

General Questions

1 What is your name?

2 How old are you?

3 What is your highest education level?

4 What is the name of your village?

5 What ward is this?

6 What union is this?

7 What upzilla is this?

8 How many members live in your household?

Availability

9 From what source do you get drinking water?

10 If tubewell, depth in feet?

11 What is/are your reason(s) for using your drinking water option?

12 How much drinking water do you collect per day for one person (in litres)?

13 Do you use any other sources at the moment (in the dry season)?

14 What is/are your reason(s) for using this drinking water options?

15 Do you use any other sources in the wet season?

16 What is/are your reason(s) for using this drinking water options?

17 Who collects the drinking water?

18 **Collection Time**. How long does it take in minutes to collect the water from the moment you leave the house until coming back (including walking, queuing, collecting)?

1. Very long (>60 min)
2. Long (30–60 min)
3. Medium (10–29 min)
4. Short (5–9 min)
5. Very short (<5 min)

19 **Palatability**. How much do you like or dislike the taste of your drinking water?

1. Strongly dislike
2. Dislike
3. Neutral
4. Like
5. Strongly like

20 **Palatability**. What is/are your reason(s) for liking or disliking it?

21 **Palatability**. Is there an iron taste in the water? Yes/No

22 **Palatability**. How is the saline taste of the water?

1. Very saline
2. Saline
3. Neutral
4. Not saline/fresh
5. Not saline at all/very fresh

23 **Cost**. Is your water free, or do you have to pay? Yes/No

24 **Cost** How do you feel about the cost?

1. Expensive

2. Cheap
3. Free

25. **Cost** How much taka do you pay per litre?

26 **Health**. How healthy do you think your drinking water is?

1. Very unhealthy
2. Unhealthy
3. Neutral
4. Healthy
5. Very healthy

27 **Health**. Why do you think it is healthy/unhealthy?

28 **Arsenic**. Do you know about arsenicosis? Yes/No

29 **Arsenic**. What are the symptoms of arsenicosis?

30 **Arsenic**. How high or low do you think is the risk that you will develop arsenicosis?

1. Large risk
2. Some Risk
3. Neutral
4. No risk

31 **Neighbours' opinion**. What do your neighbours say about your drinking water?

1. Strongly disapprove
2. Disapprove
3. Neutral
4. Approve
5. Strongly approve

32 **Regular convention** How many people from your community get water from your drinking water source?

1. Few people/less than 10
2. Intermediate amount of people/between 10 and 100
3. Many people/more than 100

33 **Habit**. Do you use your drinking water option out of habit?

1. Very unsure
2. Unsure
3. Neutral
4. Sure
5. Very sure

34 **Habit**. How long have you been drinking the water from this source?

35 **Habit**. What did you use before?

36 **Reliability**. Do you think you can get water from this source next month?

1. Very unsure
2. Unsure
3. Neutral
4. Sure
5. Very sure

37**Reliability**. Does your water source sometimes break down or becomes unavailable?

38 **Reliability**. What do you do if the water source breaks down or is unavailable?

39 **Reliability**. How often does it break down per month?

40 Water test for salinity (Electrical Conductivity) and temperature

REFERENCES

Alam, M., Sultana, M., Nair, G.B., Sack, R.B., Sack, D. a, Siddique, a K., Ali, A., Huq, A., Colwell, R.R., 2006. Toxigenic Vibrio cholerae in the Aquatic Environment of Mathbaria, Bangladesh. Appl. Environ. Microbiol. 72, 2849–2855. https://doi.org/10.1128/AEM.72.4.2849-2855.2006

Altherr, A.M., Mosler, H.J., Tobias, R., Butera, F., 2008. Attitudinal and relational factors predicting the use of solar water disinfection: A field study in nicaragua. Heal. Educ. Behav. 35, 207–220. https://doi.org/10.1177/1090198106288251

Ayers, J.C., Goodbred, S., George, G., Fry, D., Benneyworth, L., Hornberger, G., Roy, K., Karim, M.R., Akter, F., 2016. Sources of salinity and arsenic in groundwater in southwest Bangladesh. Geochem. Trans. 17, 4. https://doi.org/10.1186/s12932-016-0036-6

BGS, DPHE, 2001. Arsenic contamination of groundwater in Bangladesh; BGS Technical Report WC/00/19. British Geological Survey, Keyworth.

Bhuiyan, M.A.H., Rakib, M.A., Dampare, S.B., Ganyaglo, S., Suzuki, S., 2011. Surface water quality assessment in the central part of Bangladesh using multivariate analysis. KSCE J. Civ. Eng. 15, 995–1003. https://doi.org/10.1007/s12205-011-1079-y

Chowdhury, N.T., 2010. Water management in Bangladesh: an analytical review. Water Policy 12, 32. https://doi.org/10.2166/wp.2009.112

Despins, C., Farahbakhsh, K., Leidl, C., 2009. Assessment of rainwater quality from rainwater harvesting systems in Ontario, Canada. J. Water Supply Res. Technol. - AQUA 58, 117–134. https://doi.org/10.2166/aqua.2009.013

Dobrowsky, P.H., De Kwaadsteniet, M., Cloete, T.E., Khan, W., 2014. Distribution of Indigenous Bacterial Pathogens and Potential Pathogens Associated with Roof-Harvested Rainwater. Appl. Environ. Microbiol. 80, 2307–2316. https://doi.org/10.1128/aem.04130-13

Du Preez, M., McGuigan, K.G., Conroy, R.M., 2010. Solar disinfection of drinking water in the prevention of dysentery in South African children aged under 5 years: The role of participant motivation. Environ. Sci. Technol. 44, 8744–8749. https://doi.org/10.1021/es103328j

Dunn, O.J., 1964. Multiple Comparisons Using Rank Sums. Technometrics 6, 241–252. https://doi.org/10.1080/00401706.1964.10490181

Faisal, I.M., Kabir, M.R., 2005. An Analysis of Gender–Water Nexus in Rural Bangladesh. J. Dev. Soc. 21, 175–194. https://doi.org/10.1177/0169796X05054623

Flanagan, S., Johnston, R., Zheng, Y., 2012. Arsenic in tube well water in Bangladesh: health and economic impacts and implications for arsenic mitigation. Bull. World Health Organ. 90, 839–846. https://doi.org/10.2471/BLT.11.101253

Gaus, I., Kinniburgh, D.G., Talbot, J.C., Webster, R., 2003. Geostatistical analysis of arsenic concentration in groundwater in Bangladesh using disjunctive kriging. Environ. Geol. 44, 939–948. https://doi.org/10.1007/s00254-003-0837-7

George, C.M., Inauen, J., Perin, J., Tighe, J., Hasan, K., Zheng, Y., 2017. Behavioral Determinants of Switching to Arsenic-Safe Water Wells: An Analysis of a Randomized Controlled Trial of Health Education Interventions Coupled

With Water Arsenic Testing. Heal. Educ. Behav. 44, 92–102. https://doi.org/10.1177/1090198116637604

Graf, J., Meierhofer, R., Wegelin, M., Mosler, H.J., 2008. Water disinfection and hygiene behaviour in an urban slum in Kenya: Impact on childhood diarrhoea and influence of beliefs. Int. J. Environ. Health Res. 18, 335–355. https://doi.org/10.1080/09603120801966050

Harun, M.A.Y.A., Kabir, G.M.M., 2012. Evaluating pond sand filter as sustainable drinking water supplier in the Southwest coastal region of Bangladesh. Appl. Water Sci. 3, 161–166. https://doi.org/10.1007/s13201-012-0069-7

Harvey, C.F., Swartz, C.H., Badruzzaman, A.B.M., Keon-Blute, N., Yu, W., Ali, M.A., Jay, J., Beckie, R., Niedan, V., Brabander, D., Oates, P.M., Ashfaque, K.N., Islam, S., Hemond, H.F., Ahmed, M.F., 2002. Arsenic Mobility and Groundwater Extraction in Bangladesh. Science (80-.). 298, 1602–1606. https://doi.org/10.1126/science.1076978 1.

Hasan, M.B.; Driessen, P.P.J.; Majumder, S.; Zoomers, A.; Van Laerhoven, F., 2019. Factors Affecting Consumption of Water from a Newly Introduced Safe Drinking Water System : The Case of Managed Aquifer Recharge (MAR) Systems in Bangladesh. Water. 11, 2459.

Heri, S., Mosler, H.J., 2008. Factors affecting the diffusion of solar water disinfection: A field study in Bolivia. Heal. Educ. Behav. 35, 541–560. https://doi.org/10.1177/1090198108321248

Hoque, B.A., Hoque, M.M., Ahmed, T., Islam, S., Azad, A.K., Ali, N., Hossain, M., Hossain, M.S., 2004. Demand-based water options for arsenic mitigation: An experience from rural Bangladesh. Public Health 118, 70–77. https://doi.org/10.1016/S0033-3506(03)00135-5

Hossain, M.; Rahman, S.N.; Bhattacharya, P.; Jacks, G.; Saha, R.; Rahman, M., 2015. Sustainability of arsenic mitigation interventions-an evaluation of different alternative safe drinking water options provided in Matlab, an arsenic hot spot in Bangladesh. Front. Environ. Sci., 3, 1–15.

Huber, A.C., Bhend, S., Mosler, H.J., 2012. Determinants of exclusive consumption of fluoride-free water: A cross-sectional household study in rural Ethiopia. J. Public Heal. 20, 269–278. https://doi.org/10.1007/s10389-011-0445-z

Hurlimann, A., Dolnicar, S., 2016. Public acceptance and perceptions of alternative water sources: A comparative study in nine locations. Int. J. Water Resour. Dev. 32, 650–673. https://doi.org/10.1080/07900627.2016.1143350

Inauen, J., Hossain, M.M., Johnston, R.B., Mosler, H.J., 2013. Acceptance and Use of Eight Arsenic-Safe Drinking Water Options in Bangladesh. PLoS One 8. https://doi.org/10.1371/journal.pone.0053640

Islam, M.A., Sakakibara, H., Karim, M.R., Sekine, M., 2013. Potable water scarcity: options and issues in the coastal areas of Bangladesh. J. Water Health 11, 532–542. https://doi.org/10.2166/wh.2013.215

Kabir, A., Howard, G., 2007. Sustainability of arsenic mitigation in Bangladesh: Results of a functionality survey. Int. J. Environ. Health Res. 17, 207–218. https://doi.org/10.1080/09603120701254904

Kamruzzaman, A.K.M., Ahmed, F., 2006. Study of Performence of Existing Pond Sand Filters in Different Parts of Bangladesh. Sustain. Dev. water Resour. water supply Environ. Sanit. Proc. 32nd WEDC Int. Conf. 377–380.

Khan, A.E., Scheelbeek, P.F.D., Shilpi, A.B., Chan, Q., Mojumder, S.K., Rahman, A., Haines, A., Vineis, P., 2014. Salinity in Drinking Water and the Risk of (Pre)Eclampsia and Gestational Hypertension in Coastal Bangladesh: A Case-Control Study. PLoS One 9, e108715. https://doi.org/10.1371/journal.pone.0108715

Kjellén, M., McGranahan, G., 2006. Informal Water Vendors And the Urban Poor. International Institute for Environment and Development, London.

Knappett, P.S.K., Escamilla, V., Layton, A., McKay, L.D., Emch, M., Williams, D.E., Huq, R., Alam, J., Farhana, L., Mailloux, B.J., Ferguson, A., Sayler, G.S., Ahmed, K.M., van Geen, A., 2011. Impact of population and latrines on fecal contamination of ponds in rural Bangladesh. Sci. Total Environ. 409, 3174–3182. https://doi.org/10.1016/j.scitotenv.2011.04.043

Kruskal, W.H., Wallis, W.A., 1952. Use of Ranks in One-Criterion Variance Analysis. J. Am. Stat. Assoc. 47, 583–621. https://doi.org/10.1080/01621459.1952.10483441

Matell, M.S., Jacoby, J., 1971. Is there an optimal number of alternatives for likert scale items? study 1: Reliability and validity. Educ. Psychol. Meas. https://doi.org/10.1177/001316447103100307

Mäusezahl, D., Christen, A., Pacheco, G.D., Tellez, F.A., Iriarte, M., Zapata, M.E., Cevallos, M., Hattendorf, J., Cattaneo, M.D., Arnold, B., Smith, T.A., Colford, J.M., 2009. Solar drinking water disinfection (SODIS) to reduce childhood diarrhoea in rural Bolivia: A cluster-randomized, controlled trial. PLoS Med. 6. https://doi.org/10.1371/journal.pmed.1000125

Moser, S., Mosler, H.J., 2008. Differences in influence patterns between groups predicting the adoption of a solar disinfection technology for drinking water in Bolivia. Soc. Sci. Med. 67, 497–504. https://doi.org/10.1016/j.socscimed.2008.04.002

Mosler, H.J., Blöchliger, O.R., Inauen, J., 2010. Personal, social, and situational factors influencing the consumption of drinking water from arsenic-safe deep tubewells in Bangladesh. J. Environ. Manage. 91, 1316–1323. https://doi.org/10.1016/j.jenvman.2010.02.012

Mosler, H., 2012. A systematic approach to behavior change interventions for the water and sanitation sector in developing countries : a conceptual model , a review , and a guideline 3123. https://doi.org/10.1080/09603123.2011.650156

Naus, F.L., Schot, P., Ahmed, K.M., Griffioen, J., 2019. Influence of landscape features on the large variation of shallow groundwater salinity in southwestern Bangladesh. J. Hydrol. X 5. https://doi.org/10.1016/j.hydroa.2019.100043

Naus, F.L., Schot, P., Groen, K., Ahmed, K.M., Griffioen, J., 2019. Groundwater salinity variation in Upazila Assasuni (southwestern Bangladesh), as steered by surface clay layer thickness, relative elevation and present-day land use. Hydrol. Earth Syst. Sci. 23, 1431–1451. https://doi.org/10.5194/hess-23-1431-2019

Nickson, R., McArthur, J., Burgess, W., Ahmed, K.M., Ravenscroft, P., Rahmanñ, M., 1998. Arsenic poisoning of Bangladesh groundwater. Nature 395, 338–338. https://doi.org/10.1038/26387

Opar, A., Pfaff, A., Seddique, A.A., Ahmed, K.M., Graziano, J.H., van Geen, A., 2007. Responses of 6500 households to arsenic mitigation in Araihazar, Bangladesh. Heal. Place 13, 164–172. https://doi.org/10.1016/j.healthplace.2005.11.004

Peal, A.J., Evans, B.E., van der Voorden, C., 2010. Hygiene and Sanitation Software: An Overview of Approaches, Water Supply and Sanitation Collborative Council.

Peters, C.N., Baroud, H., Hornberger, G.M., 2019. Multicriteria Decision Analysis of Drinking Water Source Selection in Southwestern Bangladesh. J. Water Resour. Plan. Manag. 145, 05019004. https://doi.org/10.1061/(ASCE)WR.1943-5452.0001029

Sharma, B., Amarasinghe, U., Xueliang, C., de Condappa, D., Shah, T., Mukherji, A., Bharati, L., Ambili, G., Qureshi, A., Pant, D., Xenarios, S., Singh, R., Smakhtin, V., 2010. The indus and the ganges: River basins under extreme pressure. Water Int. 35, 493–521. https://doi.org/10.1080/02508060.2010.512996

Sirajul Islam, M., Brooks, A., Kabir, M.S., Jahid, I.K., Shafiqul Islam, M., Goswami, D., Nair, G.B., Larson, C., Yukiko, W., Luby, S., 2007. Faecal contamination of drinking water sources of Dhaka city during the 2004 flood in Bangladesh and use of disinfectants for water treatment. J. Appl. Microbiol. 103, 80–87. https://doi.org/10.1111/j.1365-2672.2006.03234.x

Sultana, S., Ahmed, K.M., Mahted-Ul-Alam, S.M., Hasan, M., Tuinhof, A., Ghosh, S.K., Rahman, M.S., Ravenscroft, P., Zheng, Y., 2014. Low-Cost Aquifer Storage and Recovery: Implications for Improving Drinking Water Access for Rural Communities in Coastal Bangladesh. J. Hydrol. Eng. 20. https://doi.org/10.1061/(ASCE)HE

Worland, S.C., Hornberger, G.M., Goodbred, S.L., 2015. Source, transport, and evolution of saline groundwater in a shallow Holocene aquifer on the tidal deltaplain of southwest Bangladesh. Water Resour. Res. 51, 5791–5805. https://doi.org/10.1002/2014WR016262


unicef
মার ফিল্টার (MAR Filter)
মিষ্টি পানি সারা বছর
সাইটের নাম ঃ হুলা মন্দির
গ্রাম ঃ কায়েমখোলাহুলা, ইউনিয়নঃ গঙ্গারামপুর
উপজেলাঃ বটিয়াঘাটা, জেলাঃ খুলনা
ব্যবস্থাপনায় ঃ হুলা মার ফিল্টার ব্যবহারকারী কমিটি
সহযোগিতায় ঃ লোকজ

CHAPTER 5

POTENTIAL FOR MANAGED AQUIFER RECHARGE IN SOUTHWESTERN BANGLADESH BASED ON SOCIAL NECESSITY AND TECHNICAL SUITABILITY

Floris Loys Naus

Paul Schot

Boris M. van Breukelen

Kazi Matin Ahmed

Jasper Griffioen

2020. Hydrogeology journal, Accepted for publication

ABSTRACT

In southwestern Bangladesh, scarcity of clean drinking water results in people consuming groundwater with salinity and arsenic levels exceeding drinking water standards, bacterially polluted fresh surface water and, during the monsoon, rainwater. Managed Aquifer Recharge (MAR) might potentially provide safe drinking water by storing abundant fresh surface water from the wet season in fresh-brackish aquifers for year-round use, especially during the dry season. We determined potential for MAR throughout the region by combining assessments of its necessity and the technical suitability for MAR. Assessment was based on the largest groundwater quality dataset compiled to date in southwestern Bangladesh (Khulna, Satkhira and Bagerhat districts), which contains 3716 salinity measurements and 827 arsenic measurements. The regional technical suitability assessment (1) determined the impact of density-driven flow on fresh recovery efficiency by the MAR systems in brackish groundwater environments, and (2) assessed the vulnerability of recovered water to mixing with the native brackish-saline or arsenic-contaminated groundwater. The spatial distributions of the technical suitability index and the necessity for MAR were combined to evaluate the potential for MAR throughout the region. The results show that frequently inundated areas (containing aquaculture and (tidal) river floodplains) largely limit areas where MAR systems can be installed. Elsewhere, there is a general mismatch between necessity and technical suitability. In some northern areas, necessity is low because good quality groundwater is present and hence despite their high technical suitability, potential for MAR is reduced. In other northern areas, unsafe arsenic or brackish groundwater is likely to be used for drinking water. There, MAR is a technically suitable and safer option. In southern areas, where saline groundwater is widespread and people therefore consume bacterially unsafe pond water, there is high necessity for MAR as a safe drinking water option. However, the high groundwater salinity calls for careful evaluation of MAR design and for MAR systems with a high infiltration rate to limit impacts of density-driven flow and MAR water quality deterioration. The calculated density-driven flow can be translated into a practical guideline for the approximate MAR infiltration rate needed to achieve sufficiently high recovery efficiencies. The approach developed may be useful for mapping integrated social-technical MAR potential in other saline deltas worldwide.

5.1 INTRODUCTION

Managed Aquifer Recharge (MAR) is a valuable technology for the provision of drinking water. It has been applied in different areas of the world to provide year-round drinking water access and to improve the drinking water quality (Dillon, 2005; Maliva et al., 2006; Sprenger et al. 2017; Stefan and Ansems, 2018). In southwestern Bangladesh, the provision of safe drinking water is not guaranteed, leading to the consumption of bacterially contaminated pond water (Knappett et al. 2011) and of shallow groundwater with potentially high levels of arsenic and salinity (Ayers et al., 2016; Harvey et al., 2002; Naus et al., 2019a; Naus et al., 2019b; Nickson et al., 1998; Flanagan et al., 2012; Khan et al., 2014). Overcoming consumption from these unsafe drinking water sources is directly linked to achieving sustainable Development Goal 6, to 'Ensure availability and sustainable management of water and sanitation for all' (UN, 2015). Efforts to achieve this goal in southwestern Bangladesh (Satkhira, Khulna and Bagerhat districts) have been made by piloting MAR in the form of 99 small-scale community-run systems that use sand-filtered pond water as source. The MAR systems continuously infiltrate and abstract water, with infiltration peaking in the monsoon and abstraction peaking in the dry season (Sultana et al., 2014). To develop a regional implementation strategy for MAR, an evaluation of the potential for MAR throughout the region is required.

MAR suitability mapping or MAR site selection has previously been performed in various regions in the world with a focus on assessing the technical suitability of locations (Rahman et al., 2012; Ghayoumian et al., 2007; Kallali et al., 2007; Brown et al., 2005; Chowdhury et al., 2009; Russo et al., 2015; Zuurbier et al., 2013). Most of these studies assessed the potential applicability of MAR in terms of potential for spreading basins by examining natural conditions, including surface characteristics such as soil permeability, land cover and surface slope, as well as subsurface conditions such as aquifer transmissivity and storage capacity, and the quality of the native groundwater (Rahman et al., 2012; Ghayoumian et al., 2007; Brown et al., 2005; Chowdhury et al., 2009; Kallali et al., 2007; Russo et al., 2015). In some cases, other parameters were included: for example, parameters related to economic viability, such as distance to the water source (Kallali et al., 2007). Standard practice for the site selection is to use expert judgement to interpret and combine the chosen parameters for an estimation of the site's suitability for MAR (Rahman et al., 2012; Ghayoumian et al., 2007; Kallali et al., 2007; Brown et al., 2005; Chowdhury et al., 2009; Russo et al., 2015).

In southwestern Bangladesh, the subsurface is characterised by a clayey confining top layer overlying sandy aquifers (Mukherjee et al., 2009). As a consequence, MAR systems must use injection wells, and though surface conditions limit where MAR systems can be sited, they do not necessarily control their performance. Instead, the performance is mostly determined by hydrogeological conditions, such as groundwater quality and groundwater flow velocity. Technical suitability mapping for MAR systems that utilise injection and extraction wells calls for an assessment based on these aforementioned hydrogeological conditions, and has only rarely been conducted (Zuurbier et al., 2013). Moreover, such an assessment has only rarely been validated by comparing the assessment results with actual results of implemented MAR systems (Zuurbier et al., 2013).

Previous MAR suitability studies have assumed MAR is necessary. However, it is simplistic to assume that MAR is necessary throughout southwestern Bangladesh, as illustrated by a study of Peters et al. (2019). They showed that people may prefer the drinking water options already available (like rainwater harvesting, pond sand filters, and tube wells) rather than the new MAR systems, even though the existing options have some limitations (they are not always available throughout the year or throughout the region). Therefore, to determine the necessity for MAR, it is essential to assess its benefits compared to existing water options.

The sites for the 99 pilot MAR systems in southwestern Bangladesh were selected on a case-by-case basis, based on the occurrence of prerequisites such as a suitable source for infiltration water, the lack of nearby tube wells tapping fresh groundwater, and the presence of a suitable aquifer confirmed by test drillings (Acacia 2014a; Hasan et al., 2018). On a regional scale, potential for MAR systems remains largely unknown. We therefore set out to comprehensively evaluate the MAR potential at a regional scale in southwestern Bangladesh by combining 1) an assessment of the spatially varying necessity for MAR, and 2) an assessment of the spatially varying technical suitability for MAR.

5.2 THEORY AND METHODS

Our approach to determine MAR potential comprises the combination of two assessments. Figure 5.1 shows our approach schematically. The first assessment focussed on the spatially varying necessity for MAR by examining limitations to the currently available drinking water options throughout the region, including the lack of good quality groundwater. The second assessment focussed on the technical suitability of the region for MAR systems, investigating constraints for the construction of MAR systems and using data on hydrological conditions to calculate the expected technical performance of the MAR systems. We combined these two assessments using GIS overlays to reveal the potential for MAR systems throughout the southwestern region of Bangladesh. As starting point for these analyses, we mapped the existing groundwater quality on salinity and arsenic in southwest Bangladesh, using the considerable volume of groundwater quality data that has become available in recent years (Ayers et al., 2016; Naus et al., 2019a; Naus et al., 2019b). Below, we first describe the regional groundwater quality.

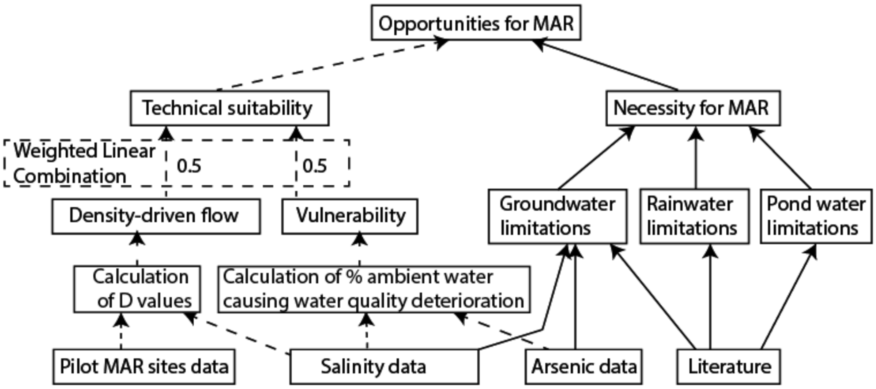


Figure 5.1. Framework of the methodology of the study to determine potential for MAR. The figure shows how the two assessments were performed and combined. D is a dimensionless value used as an indication for the amount of density driven flow (Bakker, 2010).

5.2.1 Regional characterisation of the groundwater quality

5.2.1.1 Data collection and data mining

To characterise the groundwater quality in terms of salinity, we compiled a comprehensive database from data from previous studies and unpublished data we had collected during multiple fieldwork campaigns between November 2015 and September 2018 further described in Naus et al. (2019a) and Naus et al. (2019b). For salinity, we used 2440 shallow (< 60 m deep) electrical conductivity (EC) measurements from the database of Naus et al. (2019b), supplemented with 183 intermediate deep (60-100 m) and 715 deep (> 100 m) previously unpublished groundwater EC measurements, and with 106 intermediate deep (60-100 m) and 72 deep (> 100 m) groundwater EC measurements from the Bangladesh Water Development Board (BWDB)

(2013) and from a dataset received from Vanderbilt University. Additionally, we converted sodium (Na) data from the British Geological Survey (BGS) (BGS and DPHE, 2001) into EC values, mainly to fill in the areas with few EC measurements in the northern parts of the region. For the conversion, we first assumed the concentration of Na to be about equal to Cl on a molar basis, and then converted to EC based on the correlations from Naus et al. (2019b), i.e. EC (mS/cm) = 0.0022 * Cl (mg/L) + 1.1341 (R^2 = 0.842). This yielded an additional 146 shallow, 14 intermediate deep and 40 deep data points for salinity in the region, resulting in totals of 2586 shallow, 303 intermediate deep and 827 deep groundwater data points.

For arsenic, we constructed our database from data from previous studies (Ayers et al., 2016; British Geological Survey (BGS) and Department of Public Health Engineering (DPHE), 2001; BWDB, 2013) and from data collected during the fieldwork campaigns described in Naus et al. (2019a) and Naus et al. (2019b). During the fieldwork campaigns, groundwater arsenic was measured with Hach field test kits (Hach company, USA) at 136 locations and at 295 locations by taking samples after filtering through a 0.45 μm membrane and storing them in 15 ml polyethylene tubes for analysis in the lab with an ICP-MS. In total, the arsenic database consisted of 812 shallow (< 60 m) and 52 intermediate deep (60–100 m) measurements.

5.2.1.2 Kriging

To arrive at continuous regional groundwater quality maps, we applied ordinary Kriging interpolation of the salinity and arsenic data. For the Kriging, we used a spherical variogram model that took the 8 nearest measurements into account, which, after trial and error, provided a good balance between sensitivity to local variation and sensitivity to data further away. The resulting grid consisted of cells of 30 by 30 m. The Kriging maps have as advantage that they provide a regional overview of the water quality data, but as disadvantage that they do not yield accurate predictions of the local variation in groundwater quality. For the technical suitability assessment, we only considered groundwater within manual drilling range (Acacia, 2014a), so we applied Kriging for arsenic and salinity on the data for shallow groundwater (0–60 m).

For the necessity assessment, the shallow to intermediate deep groundwater (60–100 m deep) and the deep groundwater (> 100 m) are also relevant, as both shallow and deep tube wells may be employed for drinking water supply. We therefore applied Kriging to the salinity data for intermediate deep groundwater (60–100 m) and separately for deep groundwater (> 100 m). Unfortunately, we did not have enough data for arsenic to be able to apply Kriging for the shallow groundwater separately from intermediate deep groundwater, so for the hydrogeological suitability assessment we applied Kriging across the 0–100 m depth range in addition to the 0–60 m depth range. There are no large spatial differences between the arsenic Kriging maps over these two depth ranges. As few data were available on the arsenic content of the deep groundwater, Kriging for arsenic was not possible. Although arsenic may rarely occur in groundwater deeper than 100 m, it is mostly absent in deep groundwater (BGS & DPHE 2001; Fendorf et al., 2010). We therefore assumed the deep (> 100 m) groundwater was not limited by arsenic.

5.2.2 Assessment of the necessity for MAR

We deemed there to be a necessity for MAR when the currently used drinking water options are unsafe according to drinking water standards or are temporally insufficiently available or are spatially unavailable (Table 5.1). We focused on the three most frequently used drinking water sources in the region, namely rainwater, pond water and groundwater, though there may also be other less frequently used safe drinking water option. We formulated rules for mapping the necessity for MAR, based on the spatial and temporal limitations of the three main drinking water sources, which we determined by reviewing the existing literature.

Rainwater is generally accepted to be of good quality (Peters et al., 2019; Naus et al., 2020). It can be collected everywhere in the region. However, the seasonality of rainfall causes it to only be temporally available (Chowdhury, 2010; Sharma et al., 2010). Consequently, most people who use rainwater in the monsoon season resort to a different option in the dry season, such as pond water (Naus et al., 2020). Larger storage tanks could help overcome the seasonality of rainwater as a drinking water source, but storing rainwater is challenging, as the storage tanks can be bulky and costly, and the quality of the stored rainwater can deteriorate over time (Islam et al., 2010; Despins et al., 2009; Dobrowsky et al., 2014). We therefore assumed that the current usage of rainwater is only feasible for the local population in the wet season, and that there is still a need for MAR in the dry season.

Pond water is the traditional drinking water option in Bangladesh and is available throughout the region (Kränzlin, 2000). The ponds are recharged by rainwater during the monsoon, but the use of pond water is only seasonally limited for the ponds that run dry during the dry season. Pond water is considered to be unsafe as it is very prone to bacterial and pathogenic contamination (Knappett et al., 2011). The pond water is sometimes filtered through a Pond Sand Filter (PSF) with a typical length of 2.7 m (Yokota et al., 2001), but this filtering has been found to reduce faecal coliforms and E. coli by only 75% (Islam et al., 2011). Users of pond water generally do not perceive their drinking water option very favourably (Naus et al., 2020). We consider consumption of unsafe pond water to bring a large risk of health problems, so we assumed that when pond water is used as a source for drinking water, the necessity for MAR is high.

Groundwater was introduced as drinking water source in Bangladesh in the 1960s to prevent pond water being used as drinking water. However, in the 1990s it was found that in large parts of Bangladesh, including the southwest, groundwater is contaminated with arsenic to a depth of approximately 100 m (Nickson et al., 1998; Harvey et al., 2002; BGS & DPHE 2001; Fendorf et al., 2010). In southwestern Bangladesh, both the shallow and deep groundwater can be brackish or saline (Ayers et al., 2016; Rahman et al., 2018; Naus et al., 2019a; Naus et al., 2019b). Recently, it was found that there are some areas where hydrogeological conditions result in occurrences of fresh groundwater (Naus et al., 2019a; Naus et al., 2019b). We consider groundwater constitutes a safe drinking water option throughout the year when the quality is within the Bangladesh drinking water standards (EC < 2 mS/cm, As < 0.05 mg/l) (Ayers et al., 2016).

Based on the foregoing, we distinguished three necessity classes:

1. We consider there to be **no necessity** for MAR when either shallow or deep groundwater is of good quality according to the Bangladesh drinking water standards (EC < 2 mS/cm, As < 0.05 mg/l) (Ayers et al., 2016). Deep tube wells are predominately placed on a communal level, so we assume they are only placed when the deep groundwater quality meets the drinking water standards.

2. We considered there to be medium **necessity** for MAR when unsafe groundwater is consumed. We assume this might occur when shallow groundwater is brackish (2 < EC < 5 mS/cm) or is fresh but contains arsenic (As > 0.05 mg/l).

3. When the deep groundwater is of insufficient quality and when shallow groundwater is saline (EC > 5 mS/cm), we expect people are likely to resort to consumption of pond water. We consider the long-term effects of brackish and arsenic water (Flanagan et al., 2012; Khan et al., 2014) to

carry less of an immediate health risk than switching to pond water, as drinking pond water will likely result in more immediate health problems such as diarrhoea (Islam et al., 2011; Knappett et al., 2011). We therefore considered there to be a **high necessity** for MAR when groundwater is saline (EC > 5 mS/cm).

To map the necessity for MAR in southwestern Bangladesh, we applied the abovementioned criteria for the three necessity classes to the Kriging maps of the groundwater quality, using a GIS overlay. As described in 5.2.1, we used both the EC and arsenic for the shallow to intermediate–deep (0–100 m) groundwater quality, but for the deep (> 100 m) groundwater quality we only considered the EC.

Table 5.1. The necessity for MAR systems as a function of temporal water availability and water quality safety for the three main drinking water sources in southwest Bangladesh. PSF = pond sand filter.

	Water quality problems	Temporal availability	Spatial availability	Necessity for MAR
Pond water PSF	Bacteria, pathogens, with PSF only removing up to 70%	Some ponds dry up in dry season	Ponds are available throughout the region, with PSF at limited locations	Yes, a primary necessity, because of water quality and possible availability problems
Rainwater	-	Unavailable in dry season	Available throughout the region	Yes, because of availability problems during dry season
Groundwater	At several locations: Salinity, arsenic or both at levels above drinking water standards	-	Water quality varies spatially	Dependent on the area: No, where drinking water standards not exceeded Yes, where salinity or arsenic exceeds the drinking water standards

5.2.3 Assessment of the technical suitability

5.2.3.1 Constraint mapping

We assessed the technical suitability by filtering out constraints in the region that rule out the placement and operation of MAR systems and by calculating the expected technical performance of the MAR systems from hydrogeological conditions. For the constraint mapping, we filtered out locations where prerequisites for MAR are not met. This was done using Boolean logic, similar to the approaches by Kallali et al. (2007) and Rahman et al. (2012). The prerequisites are summarised in Table 5.2. For placement, MAR requires suitable land where infiltration and abstraction wells can be installed and operated continuously (Brown et al., 2005; Ghayoumian et al., 2007). We therefore filtered out the protected Sundarbans mangrove area and the often-inundated aquaculture ponds and tidal river floodplains, using a landcover map constructed from supervised classification of Landsat imagery.

For operation, MAR requires a sandy aquifer where water can be injected and stored, with the storage capacity, the infiltration capacity and the transmissivity of the aquifer primarily controlling the potential capacity of MAR systems (Ghayoumian et al., 2007; Brown et al., 2005; Chowdhury et al., 2009). We stipulated that the target aquifer had to be present within manual drilling range of up to 60 m deep, to keep the costs of installation acceptable (Acacia, 2014a).

In our study, we collected 875 borehole descriptions from the DPHE, BWDB, the UNICEF MAR project, Naus et al. (2019a), Naus et al. (2019b) and Ayers et al. (2016). Many of the borehole descriptions were collected by drilling using the 'sludger' or 'hand-flapper' method (Horneman et al., 2004) and, therefore lacks details on storage capacity, infiltration capacity or aquifer transmissivity. To determine whether a sufficient aquifer is present, we assumed that any material that described as sand is suitable for the MAR systems, but that such a layer had to be thicker than 9.1 m (30 ft) to obtain sufficient storage, corresponding to the thickness used as a requirement for the pilot MAR systems (Acacia, 2014a). We determined the thickness of the aquifer within the first 60 m for each borehole log, subsequently followed by spatial interpolation using Kriging, for which we used the same configuration as for the water quality data.

Table 5.2. Criteria and associated data used for the constraint mapping

Constraint criteria	Data	Corresponding value
Study area	Administrative boundaries	Satkhira, Khulna, Bagerhat districts: 1 Other districts: 0
Inundation likely or protected land	Supervised classification of cloudless Imagery from Landsat 8 (17th and 24th of March 2015, paths 137 and 138, row 44), calibrated with observations from aerial images and field observations	Treed village, one season rice, multiple season rice: 1 Water, (tidal) rivers, aquaculture, Sundarbans: 0
Aquifer	Thickness greater than 9.1 m (30 ft) within the first 60 m	Thickness less than 9.1 m (30 ft): 0 Thickness greater than 9.1 m (30 ft): 1

5.2.3.2 Technical performance

i) Theory

The performance of a MAR system that uses injection and abstraction wells is commonly assessed using the Recovery Efficiency (RE), defined as the percentage of infiltrated water that can be recovered while maintaining sufficient quality (Bakker, 2010; Maliva et al., 2006; Ward et al., 2009). The RE can be limited when the quality of the stored water is deteriorated by ambient water, which can occur due to lateral flow, diffusive/dispersive mixing and density-driven flow (Bakker, 2010; Lowry and Anderson, 2006; Maliva et al., 2006; Ward et al., 2009; Zuurbier et al., 2013).

Lateral flow is caused by the natural hydraulic gradient and leads to the infiltrated fresh water flowing away from the abstraction well. For the regional site selection, lateral flow was anticipated to be negligible, as the hydraulic gradient in the region was expected to be low because elevation differences are small and the infiltration rates are low due to the very thick confining clay layer (Naus et al., 2019a; Ayers et al., 2016). Diffusive/dispersive mixing occurs whenever there is a concentration gradient, according to Fick's law and due to aquifer anisotropy and heterogeneity during lateral flow, density-driven flow and enhanced flow from the injection wells towards the abstraction wells (Ward et al., 2009). In practice, it is impossible to predict diffusive/dispersive mixing (Ward et al., 2009), especially in southwestern Bangladesh, where data on the anisotropy or heterogeneity of the lithology are sparse. Density-driven flow is caused by differences between the density of the injected water and of the native groundwater. The more saline the surrounding groundwater, the larger the buoyant force is and the more likely it is that upconing of native, saline groundwater will reach the bottom of the abstraction well. Additionally, a larger buoyant force causes injected fresh water to be pushed to the top of the aquifer and consequently flow away laterally from the abstraction well (Bakker, 2010; Ward et al., 2009).

It is possible to assess the expected regional varying density-driven flow based on the groundwater salinity database. Whenever native groundwater ends up in the well by any of the aforementioned processes or any unforeseen processes, the quality of the abstracted water will decrease, as the primary reason for installing an MAR system in southwestern Bangladesh is that the native groundwater present is not suitable for drinking. In southwestern Bangladesh, the groundwater salinity and arsenic concentration are two major concerns for the water quality in the MAR system and the vulnerability of a MAR system to any disturbances varies according to the quality of the native groundwater that ends up in the abstraction well. This vulnerability can therefore be assessed using our groundwater salinity and arsenic database.

During our regional technical performance assessment, we assessed the expected technical performance of MAR by estimating the expected effect of density-driven flow and by estimating the vulnerability to any of the mixing processes. We combined these two variables into one technical suitability index.

ii) Density-driven flow assessment

The spatial varying magnitude of density-driven flow on the performance of MAR systems throughout the region was estimated using the method of Bakker (2010). Bakker (2010) described that the effect of density-driven flow on the RE for a MAR system typically composed of one joint injection and abstraction well depended on the duration of the injection period relative to the duration of the storage and abstraction period, and on a dimensionless parameter D which governs the flow in the MAR system. There are some important differences between the MAR systems for which the D value was developed and the MAR systems in southwestern Bangladesh in terms of their design and operation. Firstly, in the MAR systems of Bakker (2010), a single well that penetrates the entire aquifer is used for injection and abstraction, whereas the MAR systems in Bangladesh typically have 4 to 6 infiltration wells with a length of 9.1 m (30 feet) situated around one separate abstraction well of 3 m (10 feet) long that is installed approximately 3 m (10 feet) higher in the aquifer (Acacia, 2014a). Secondly, the MAR systems in Bangladesh do not have clearly demarcated injection and abstraction periods. The continuous injection and abstraction could lead to some of the water having a relatively short storage time, although on average there is a higher injection rate in the wet season and there is a higher abstraction rate in the water-stressed dry season, resulting in storage from the wet season into the dry season. Despite these differences, we used the D value developed by Bakker (2010) as previously used for MAR site selection by Zuurbier et al. (2013) as a best estimate of the spatially varying magnitude of density-driven flow on the performance of MAR systems throughout the region:

$$D = \frac{Q}{k\alpha H^2} \tag{1}$$

with Q being the infiltration and abstraction rate during the days that infiltration or abstraction occurs (m/day), k being the hydraulic conductivity, H being the thickness of the aquifer and α being the density difference ratio, calculated by:

$$\alpha = \frac{(\rho_s - \rho_f)}{\rho_f} \tag{2}$$

with ρs being the density of the native saline groundwater and ρf being the density of the injected fresh water. The higher the D value is, the higher the RE. The density of the native groundwater varies spatially and was calculated based on the electrical conductivity of the shallow groundwater (< 60 m), using the 1980 UNESCO state equation typically used for ocean water (for the full documentation, see Fofonoff and Millard 1983). Post (2012) showed these equations to be useable for calculating the density of coastal groundwater. Exact information on the hydraulic conductivity (k) was not available throughout the region, so the value of k was based on the median value of 10.9 m/day (σ: 5.68) obtained from 10 pumping tests performed by Acacia (2014b). The other parameters, H and Q, are based on the design and capacity of the pilot MAR. H was put at 9.1 m, similar to the length of the filter of the injection wells used in the pilot MAR systems (Acacia 2014a), and similar to the minimum required thickness of the aquifer, although the total thickness of the aquifer can be different than H. Q was given a value of 5 m^3/day, which is slightly less than the median injection capacity (5.9 m^3/8 hours) of the pilot MAR systems but higher than the median of the averaged actual infiltration rates per day (3.1 m^3/day) (Acacia 2014b).

To interpret an associated RE from the D value, the design, injection, storage, and recovery periods, and the amount of cycles are important (Bakker 2010). Bakker formulated how to translate D into RE for his typical design in which the injection period is of similar duration to the recovery period. Zuurbier et al. (2013), who also applied Bakker's method to assess MAR suitability, used a D value of 14.3 as criterion for well-functioning MAR systems with injection, storage and recovery periods of equal durations, related to an RE of 60% after five cycles. Due to the aforementioned differences in design and operation between the pilot MAR systems and the MAR systems of Bakker (2010) and Zuurbier et al. (2013), we could not use the same criterion. The Bangladesh MAR systems are likely to be less susceptible to density-driven flow than the MAR systems of Bakker (2010) and Zuurbier et al. (2013) because more upconing is needed for the native groundwater to reach the bottom of the separately installed abstraction well with a screen at shallower depth than the screens for the injection wells (Maliva et al., 2006; Zuurbier et al., 2014). Additionally, the simultaneous injection and abstraction in the MAR systems is likely to result in much shorter average storage times than in the MAR systems of Zuurbier et al. (2013), which will result in less time for density-driven flow and an associated higher RE for similar D values.

In view of the foregoing, we expected the design and operation differences would likely lead to a much higher RE for similar D values. As a best guess, we chose to use a D value of 5 as the approximate criterion for well-functioning MAR systems, corresponding with an approximate EC of 3.1 mS/cm. In areas with a lower D value, density-driven flow is interpreted to noticeably reduce the efficiency of the MAR system. We decided to report the D value instead of an associated RE value, as the associated RE value is subject to MAR design choices and interpretation. For guidance, we expected a D value above 5 to likely lead to an RE higher than 60% and a D value below 1 to likely lead to an RE of approximately 25% in the unmodified pilot MAR design.

iii) Vulnerability to any of the mixing processes

We expressed the vulnerability to water quality deterioration due to mixing by calculating the fraction of native water that would result in the water quality being below the water quality standards of the Bangladesh government (Ayers et al., 2016). The maximum fraction of native water (fi) in the abstracted water before a certain compound i deteriorates the abstracted water was calculated as follows:

$$f_i = \frac{i_c - i_i}{i_n - i_i} \tag{3}$$

with ic being the concentration criterion according the drinking water quality standards, ii being the concentration of the compound in the injected water, and in being the concentration of the compound in the native groundwater. The smaller the fraction, the less water is needed to deteriorate the water quality of the extracted water and the higher the vulnerability to any of the aforementioned processes. Vulnerability to deterioration was calculated for EC and arsenic, with the lowest value for these two determining when the water quality would be considered as deteriorated. For the quality of the native groundwater, we used the database of the water quality data points for groundwater down to approximately 60 m deep, as described earlier.

iv) Technical suitability index

To summarise the technical suitability of the region for MAR, we combined the expected density-driven flow in the form of the calculated D value and the vulnerability to the mixing processes in one technical suitability index. For this, we used a Weighted Linear Combination (WLC), similarly to Saraf and Choudhury (1998) and Rahman et al. (2012):

$$\text{WLC, } S(x_i) = \sum w_i \cdot s_i(x_i) \tag{4}$$

in which wi is the weight of each criterion, the sum of the weights is 1, and s_i (x_i) are the standardised criteria. We used equal weights for the expected density-driven flow and for the permitted fraction of native groundwater, i.e., w_i = 0.5.

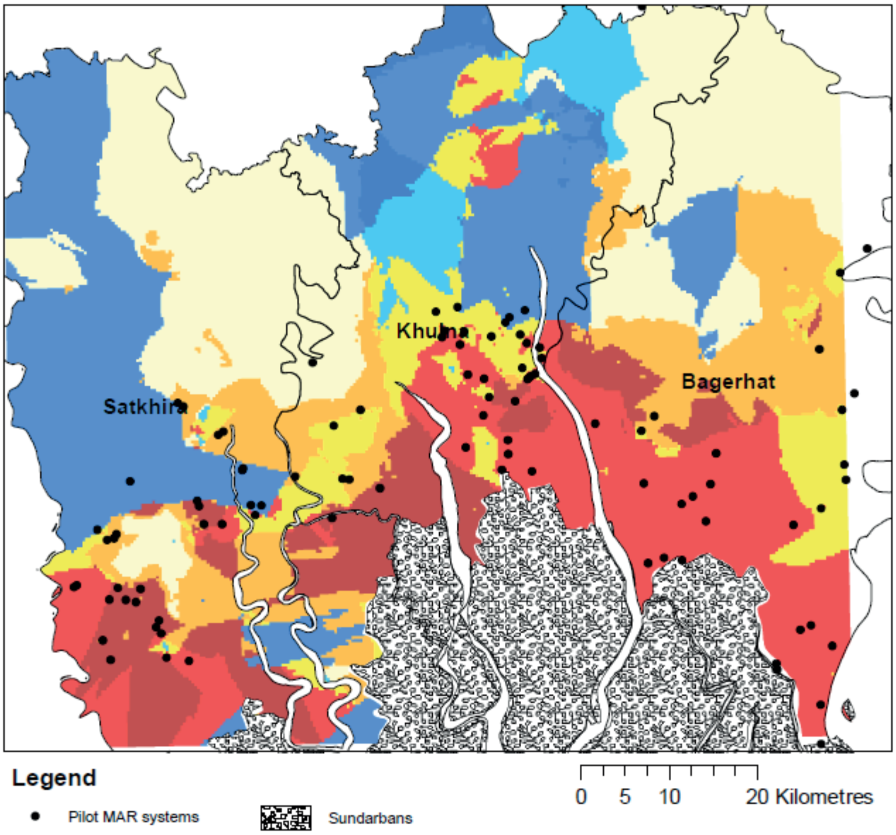
We standardised the D value and the permitted fraction of native groundwater so that they both had the same scale of 0 (no suitability) to 1 (high suitability). For standardisation of the D value, we interpreted areas with a D value below 5 as likely to be influenced by density-driven flow, corresponding to a standardised value of 1, which decreases linearly to 0 when the D value is 0. For the permitted mixing fraction, we based the standardisation on the scenarios described by Ward et al. (2009), which regularly have mixing of up to 20% native water. We standardised the potential linearly from 0 at 0% native water needed to deteriorate the water quality, to a potential of 1 when more than 30% native water is needed to deteriorate the water quality. The resulting technical suitability index is not absolute but relative. Areas with a high index are more likely to have a technical well-functioning MAR system, but areas with a low index are not guaranteed to have non-functional MAR systems.

5.3 RESULTS

We present the results of the MAR suitability mapping as follows. The results of the assessment of the necessity for MAR throughout the region are presented first, followed by the results of the technical suitability assessment. Last, the potential for MAR throughout the region is presented, combining the two assessments. The water quality database on which the assessments are based is presented as maps in the appendix.

5.3.1 Necessity for MAR

Figure 5.2 shows the spatial distribution of the necessity for MAR systems as inferred from groundwater quality. Note that the necessity applies particularly to the dry season, as rainwater is abundant during the monsoon. Larger areas in the middle/north and in the west, and smaller areas in the southwest and northeast, have groundwater of adequate quality available, indicating MAR systems are not necessary. Note that these areas mostly have deep groundwater available for use. Shallow groundwater is suitable for use as drinking water in only a few locations in the middle/north. The rest of the area has groundwater of inadequate quality because of its arsenic content and/or salinity. Although the groundwater quality in eastern Sathkira and in northern Bagerhat is mostly limited by arsenic, it is nonetheless probably consumed, stressing the necessity for safer water sources. In the southern part of the region, salinity renders the groundwater unusable. Here, we presume that consumption of surface water occurs and, therefore, there is great need for better water options.



Necessity for MAR	Shallow EC (mS/cm)	Shallow As (µg/l)	Deep EC (mS/cm)	Deep As (µg/l; assumed)
No necessity	< 2	< 50	< 2	< 50
No necessity	> 2	> 50	< 2	< 50
No necessity	< 2	< 50	> 2	< 50
Necessity	< 2	> 50	> 2	< 50
Necessity	> 2, < 5	< 50	> 2	< 50
Necessity	> 2, < 5	> 50	> 2	< 50
Primary necessity	> 5	< 50	> 2	< 50
Primary necessity	> 5	> 50	> 2	< 50

Figure 5.2. Spatial distribution of necessity for MAR as a function of observed groundwater quality, using the Bangladesh drinking water standards for salinity (EC) and arsenic (As) as indicators.

5.3.2 Technical suitability

5.3.2.1 Constraint mapping

The protected Sundarbans and the frequently inundated areas (containing aquaculture and (tidal) river floodplains) largely limit the areas where MAR systems can be installed (Figure 5.3). The Sundarbans in the south do not contain any permanent settlements, so local scale MAR systems would not be needed. The aquaculture is located in the west and southeast of the region. In the small villages between the inundated areas with aquaculture, MAR can be placed, although the land available will be limited and care should be taken to prevent lateral inflow of infiltrated saline aquaculture water.

The Kriging results of 875 borehole logs indicate that throughout the region, aquifers thicker than the formulated minimum requirement of 9.1 m (30 ft) occur within the first 60 m (Acacia, 2014a) (Figure A5.2). Nonetheless, 48 borehole logs (approximately 5%) indicate that local aquifer thickness may be less than 9.1 m. Since variation in lithology is large, this reveals that aquifer thickness cannot be accurately predicted on a regional scale, indicating the need for a test drilling when selecting sites for MAR.

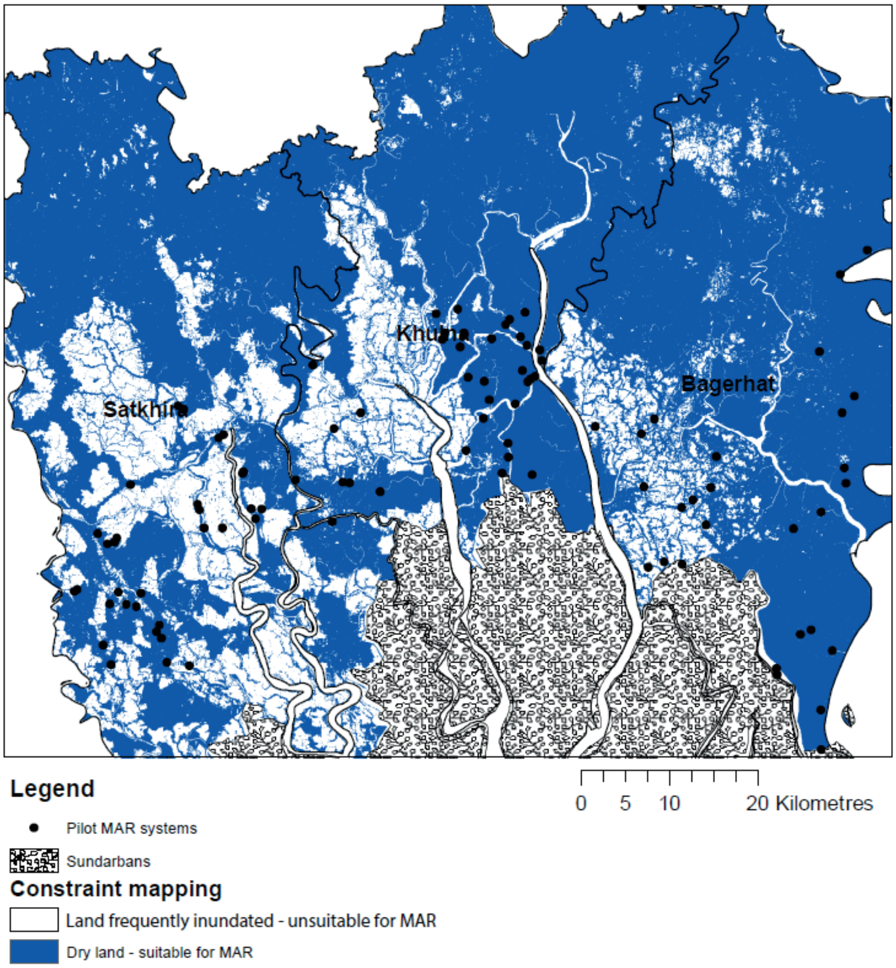


Figure 5.3. Areas considered potentially suitable for MAR, based on inundation likelihood and protected area status.

5.3.2.2. Density-driven flow

The expected influence of density-driven flow, expressed in D values, varies with the groundwater salinity patterns in the region (compare Figure 5.4 with Figure A5.1). In the northwest and northeast, the D values exceed 10. It will be recalled that we used a D value of 5 or higher, related to an RE of more than 60%, as the threshold above which density-driven flow is not expected to be an issue for the functioning of MAR systems. In large areas of the region, mainly the middle and south, the D values are below 5, indicating that the effect of density-driven flow is expected to be a reduced RE potentially affecting the feasibility of MAR. In the southwest and southeast of the region, the D values are lower than 1, which is interpreted to lead to an RE below approximately 25%, which indicates significant influences of density-driven flow on the practical application of the technology.

Since the D values were calculated using Q and H from the pilot MAR sites, they are generally relevant for predicting the effect of density-driven flow on MAR systems having the same design and infiltration regime as the pilot systems. The relatively low infiltration and abstraction rates (Q = 5) of the pilot MAR sites cause density-driven flow to be already relevant (D values < 5) in areas with a relatively low salinity (EC > 3.1 mS/cm). Indeed, the D value is lower than 5 at almost all locations where brackish groundwater is present. This indicates that either the MAR users should be satisfied with lower REs, or that improvements in the operational and design aspects should be considered.

5.3.2.3. Vulnerability map

Figure 5.5 shows the percentage at which mixing of infiltrated fresh water with native groundwater causes abstracted MAR water to fail to comply with the drinking water standards for salinity or arsenic. In this assessment, the data on shallow (0–60 m) groundwater quality were used (as shown in Figure A5.1) and potential arsenic (im)mobilisation processes were ignored. Mixing is caused by dispersion and imperfect design: for example, when the abstraction well starts to tap native groundwater while there is still infiltration water in the system. Several clear patterns are visible: areas in the northwest and northeast are more sensitive to arsenic than to salinity, while areas in the southwest and southeast and in the middle of the region are more sensitive to salinity than to arsenic. The lower the percentage of native groundwater, the higher the sensitivity to mixing, as mixing with less native groundwater would already exceed a drinking water standard (for salinity or arsenic) of the recovered water. This means that when any unforeseen mixing occurs, or when the mixing zone reaches the abstraction well filter, abstraction has to be stopped sooner, which will reduce the average RE. The susceptibility to mixing is the largest in the saline southwest and southeast, where mixing with less than 15% native groundwater will already cause the salinity standard for drinking water to be exceeded. In the northwest and the northeast, the mixing risk is still quite large even though salinity is low, because arsenic concentrations are high: 15–30% native groundwater will already cause the arsenic standard to be exceeded.

5.3.2.4. Technical suitability index

Figure 5.6 presents a map of the technical suitability index, for which the two assessments for constraint mapping – the density-driven flow effect and the vulnerability to mixing – were combined. There are clear differences in the technical suitability for MAR systems over the study region. The spatial patterns are largely dictated by the groundwater salinity patterns, due to the twofold influence of groundwater salinity on the technical suitability: it impacts both the likelihood of density-driven flow and the negative effect of mixing. This results in the overall pattern showing a decrease in technical suitability index towards the south, with the southeast having the lowest technical suitability index. However, note that areas having high groundwater arsenic concentrations such as the northeast also have a technical suitability index below 1, due to the risk for mixing with native arsenic-contaminated native groundwater.

5.3.3. Overall integrated assessment combining the necessity and technical suitability assessments

The outcome of combining the necessity assessment (Figure 5.2) and the technical suitability assessment (Figure 5.6), is presented in Figure 5.7. This combination determines the potential for MAR in the region. This potential is highest in areas where necessity and technical suitability are both high. However, the combined map shows that there are few such areas: there appears to be a mismatch between the technical suitability and the necessity for MAR. In areas with a high technical suitability index, the necessity for MAR is usually lacking or not convincing. These areas are in the northern part of the study area, where groundwater is of good quality. In contrast, in areas with a high necessity for MAR, the technical suitability index is often below 0.75 or even below 0.5. These areas are mostly in the south of the region, near the Sundarban

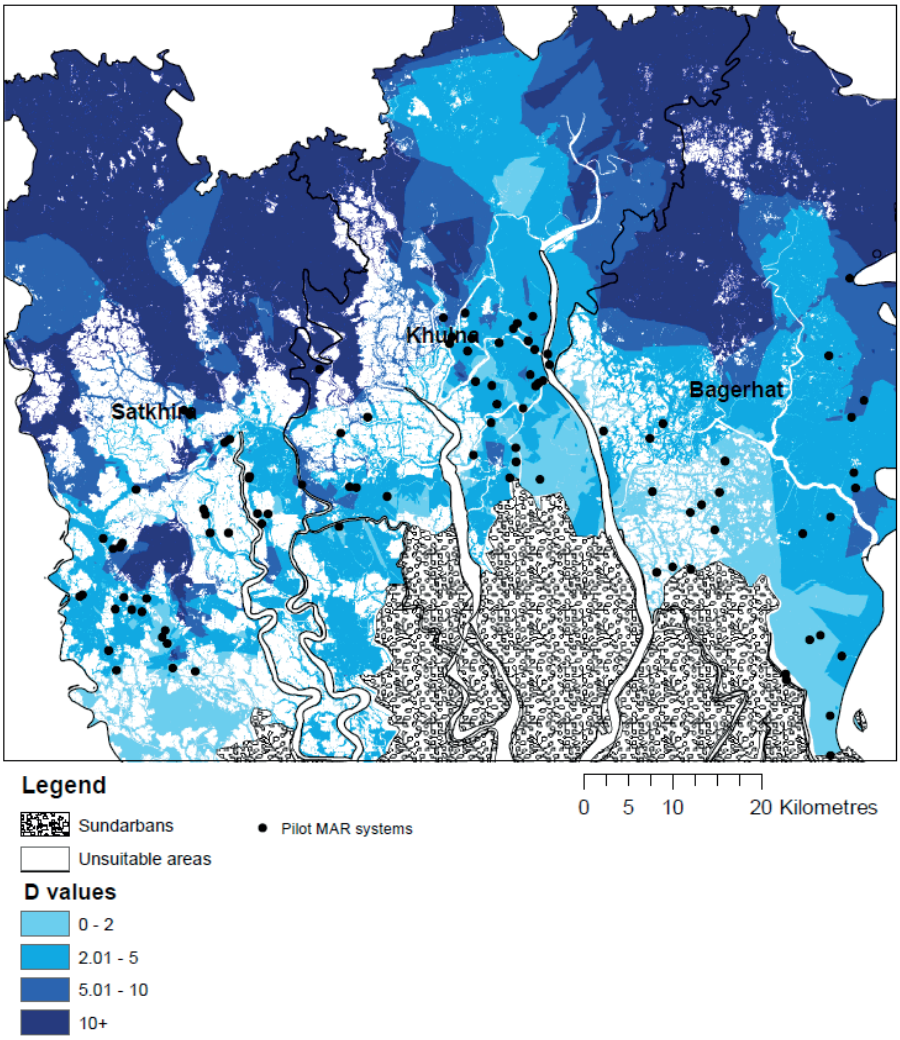


Figure 5.4. Relative influence of density-driven flow on the efficiency of MAR systems for recovering infiltrated fresh water in brackish aquifers. The lower the calculated D value, the higher the relative influence of density-driven flow.

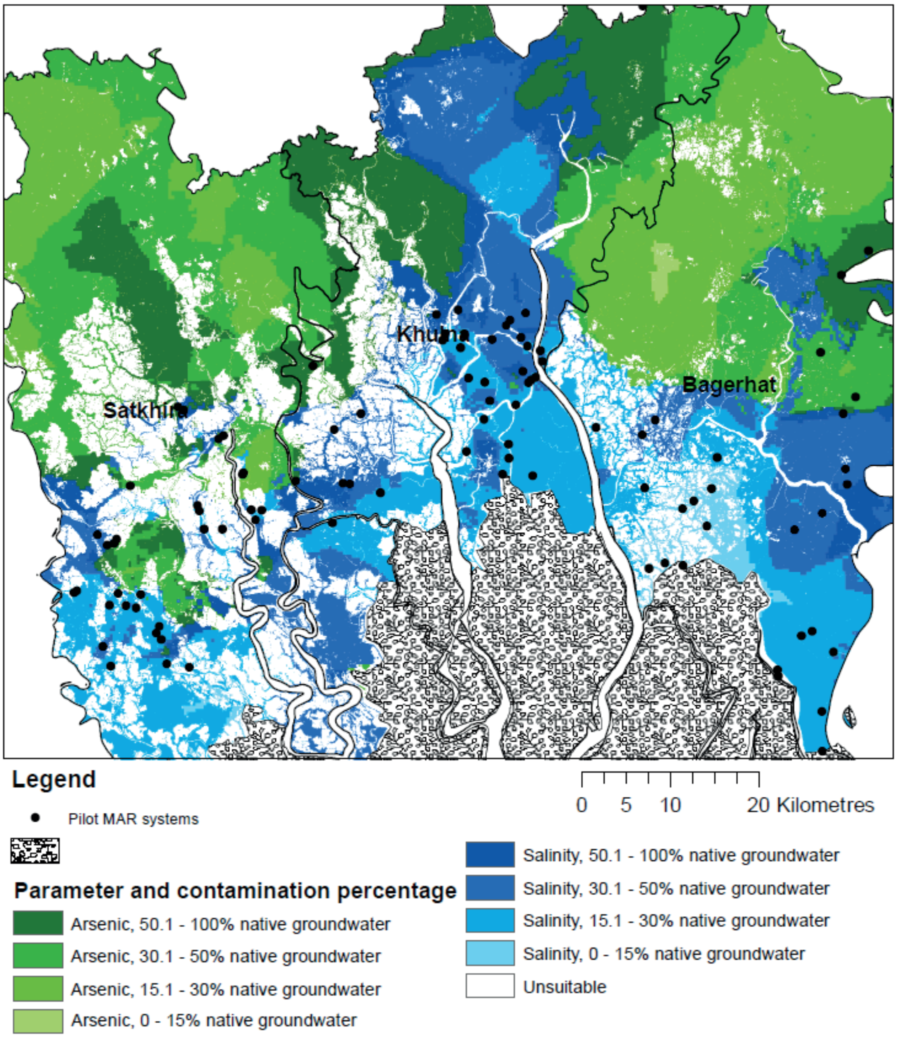


Figure 5.5. Vulnerability map that illustrates the vulnerability of MAR water to mixing with native groundwater either due to arsenic (green) or salinity (blue). The percentage of native groundwater inmixing is shown that would cause MAR water quality to exceed the Bangladesh drinking water standards. The lower the percentage, the more vulnerable the MAR system is (the lighter the colours).

5

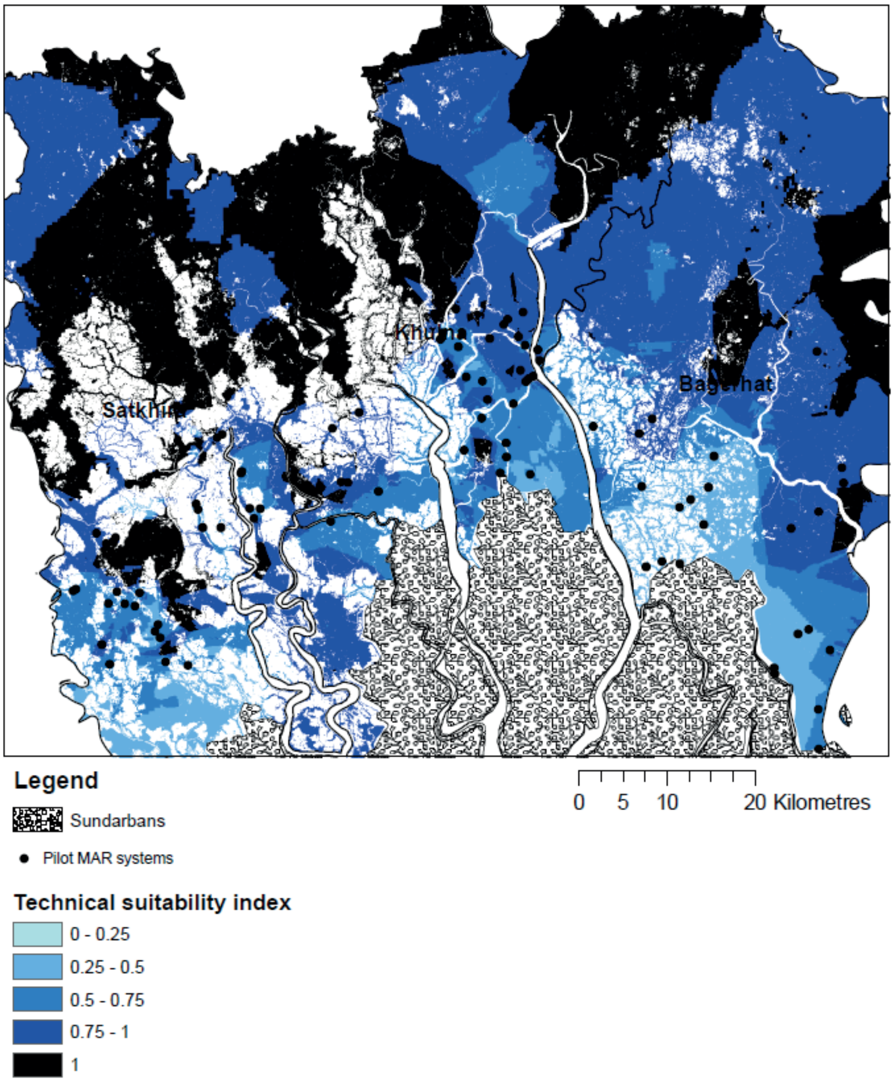


Figure 5.6. Technical suitability index for MAR throughout the study region. The lower the index value, the lower the expected technical performance of the MAR systems under the local hydrological conditions, with a higher chance of insufficient MAR water quality.

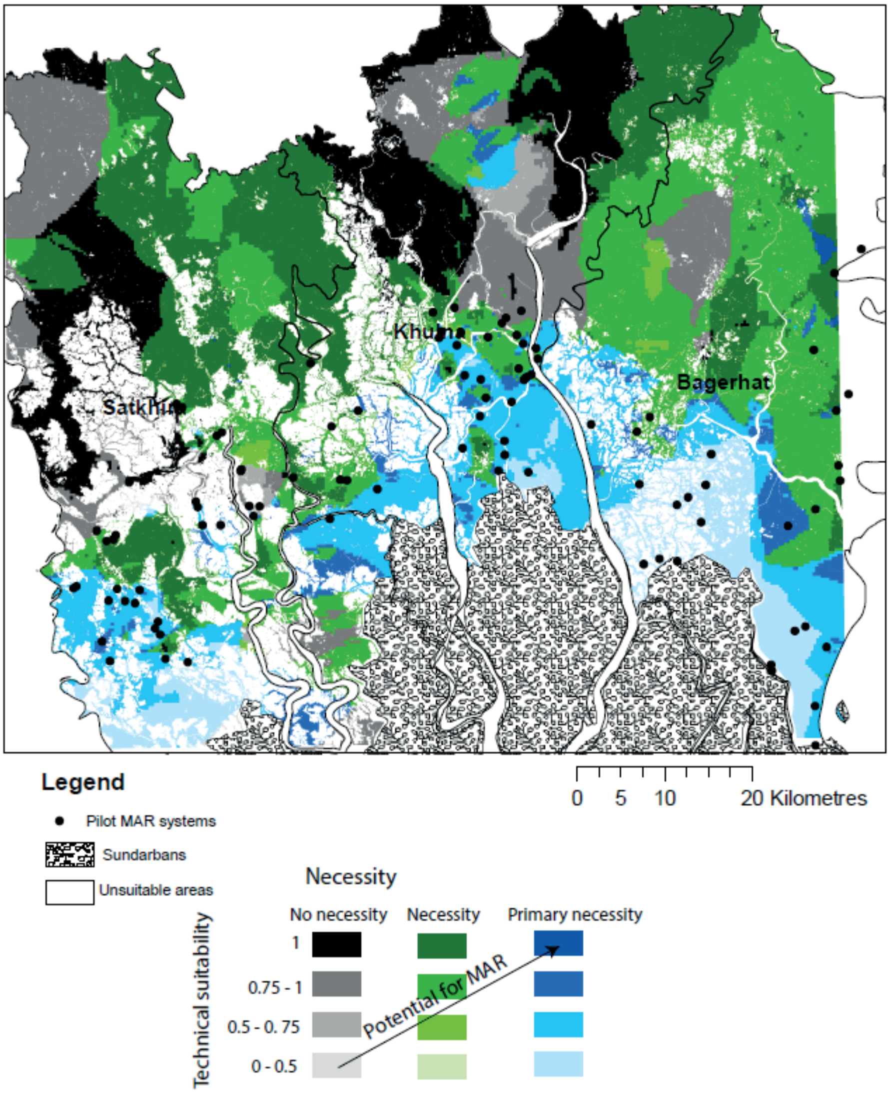


Figure 5.7. Potential for MAR based on the combination of the necessity for MAR and the technical suitability index. The higher the necessity and technical suitability index, the higher the potential for MAR.

5.4 DISCUSSION

5.4.1 Evaluation of the regional MAR potential

The potential for MAR throughout the region is controlled by both the necessity and technical suitability for MAR. The results of the necessity assessment (Figure 5.2) provide the first clear and comprehensive spatial overview of drinking water quality problems in the region, based on the most exhaustive groundwater quality database for the region to date. This reveals the areas where MAR may be applied to establish safe drinking water supply and could be used to develop a regional water resource management strategy. The technical suitability map provides valuable insights into the likelihood that MAR will function well and efficiently from a technical point of view.

In the parts of the region without a necessity for MAR, installing MAR systems may not be effective to solve current drinking water problems as the local people are not expected to be sufficiently motivated to pay for or adopt the MAR systems. Therefore, we interpret the potential for MAR in these parts to be low, even though MAR systems would be unlikely to face technical issues. However, MAR could become relevant in the future as a sustainable water option, especially when abstraction of groundwater becomes unsustainable, as has been shown in other parts of Bangladesh (Shahid and Hazarika, 2010; Hoque et al., 2007). MAR would then become valuable to possibly make unsustainable groundwater abstraction sustainable.

In the areas with a medium necessity and high technical suitability, MAR systems have potential to solve the current drinking water problems related to consumption of groundwater with unacceptable levels of arsenic or salinity. In these areas, technical complications are not expected, but the acceptance and adoption of community-run MAR systems could be problematic, as it has been found that people are attached to using their shallow tube wells as source of drinking water (Naus et al., 2020). We therefore recommend evaluating the concept of MAR systems that are solely community-run.

In areas with both a primary necessity and a low technical suitability, it is likely that community-run MAR systems will be accepted and adopted, as it has been found that people are less attached to drinking pond water (Naus et al., 2020). However, MAR systems are unlikely to function optimally. Nevertheless, MAR systems could still provide a valuable bacterially safe drinking water source if the monsoon water captured in ponds is more than sufficient to make up for losses of infiltrated fresh water resulting from non-ideal hydrogeological conditions. We therefore expect there is potential for MAR in these areas, but to achieve and maintain technically well-functioning and efficient systems in this part of the region, we recommend carefully assessing the technical MAR functioning a priori together with a targeted monitoring strategy. Below we discuss recommendations for better functioning and monitoring of MAR systems.

5.4.2 Recommendations for the monitoring of MAR systems

While the MAR systems are operating, the chemical water quality of the abstracted water should be monitored so that when the abstracted water is no longer safe for use, abstraction can be stopped. The monitoring of the pilot MAR sites stopped after a few years. It should be noted that MAR water can also be of insufficient quality if pathogens remain in the water: Sultana et al. (2014) and Kabir et al. (2016) showed that E. coli was not fully removed in some of the pilot MAR sites, so monitoring of the microbial water quality is required.

The vulnerability map (Figure 5.5) is useful for developing strategies for monitoring chemical water quality. Firstly, as the map reveals which water quality parameter (salinity or arsenic) will exceed the drinking water standard first, the map can be used to decide which parameter has priority to be monitored during operation of the MAR systems. In the south and middle of the region, salinity monitoring with EC measurements will suffice to detect the abstraction of unsafe water (on the condition that arsenic is not mobilised under site-specific conditions), while in the northwest and northeast, arsenic monitoring should have priority. It should be noted that geochemical processes with consequences for the arsenic concentration could occur: arsenic and iron may become immobilised when infiltrated aerobic water causes iron oxides to form (van Halem et al., 2010), but the reverse process may occur when organic matter remains in the infiltrated water and becomes oxidised by iron oxides, resulting in arsenic and iron mobilisation (Ravenscroft et al., 2005).

Secondly, the calculated mixing fraction is also a useful parameter to determine the frequency of monitoring: the lower the percentage of native groundwater needed to compromise the quality of the abstracted water, the more frequent the monitoring should be, especially towards the end of the dry season when little fresh water is stored underground. In the areas vulnerable to mixing with native groundwater containing arsenic, it is recommended that monitoring is also frequent, since arsenic is not detected by taste, whereas salinity is, and to account for the aforementioned geochemical processes.

5.4.3 Enhancing MAR capacity to improve performance

Note that the D value follows from given hydrogeological conditions (hydraulic conductivity, aquifer thickness, density contrast with native groundwater) and the applied infiltration/abstraction rate. D values are low over much of the region but would increase if the applied infiltration/abstraction rate is enhanced. Thus when more water is infiltrated (and abstracted) and the capacity of MAR systems is increased, the technical performance of the MAR will be better. The calculated D value can be translated into a practical guideline for the approximate MAR capacity required to achieve sufficiently high recovery efficiencies. Larger infiltration and abstraction rates (Q) are directly related to a higher D value (equation 1). Consequently, the relative loss of water due to density-driven flow can be reduced by primarily increasing the infiltration (and the abstraction) rate of the MAR systems. We calculated the Q required to limit the impact of density-driven flow and make MAR technically feasible throughout the region. To this end, we fixed the D value in equation (1) at 5 (interpreted to correspond to an expected RE of 60%), and subsequently solved for Q, similar to Zuurbier et al. (2013). We used the resulting Q values (see Figure 5.8) as a guideline for the approximate target capacity of the MAR systems required to achieve proper technical functioning. In most of the region, a D value of 5 is achieved when the infiltration and abstraction rates are doubled to 10 m^3/day. In the saline south of the region, higher infiltration and abstraction rates are required to achieve a D value of 5: up to a maximum of 30 m^3/day. To assess the feasibility of achieving these capacities, we compared them to the infiltration rate of the current pilot MAR systems. The assessed capacity of the infiltration rate of the current MAR systems varies between 1 and 19.4 m^3/day, with 5.9 m^3/day as median value, based on 8 hours of infiltration (we used 5.0 m^3/day in our assessment). This median value is lower than the Q required in a large part of the region, revealing that the current pilot MAR systems will probably not achieve a high RE. It is possible to increase the infiltration rate of the systems by increasing the hours of infiltration per day, or by installing more or larger sand filtration chambers and infiltration wells. For example, doubling or tripling the infiltration time per day and the number of infiltration wells would result in a 4 to 9 times increase in infiltration capacity, making the MAR systems feasible in almost the entire region. The relatively thin aquifers expected in the south (Figure A5.2) may limit the possibility of applying long infiltration wells.

An increase in infiltration capacity may be limited by the amount of source water available. The current MAR systems take most of their water from ponds. Data on the sizes of these ponds next to the MAR sites provided by the staff of the UNICEF MAR project revealed that pond size varies greatly: between 575 m^2 to 8261 m^2 with a median size of 1486 m^2. If only the net direct rainfall (here taken to be 2000 mm per year) to the ponds is assumed to be usable for infiltration (and not also inflow from surrounding areas), the available source water is as little as 1150 m^3 per year for the smallest ponds, and 16722 m^3 per year for the largest ponds, with a median of 2972 m^3 per year, which would allow for a infiltration rate between 6.4 and 92.9 m^3 per day during the wet season (180 days), with a median of 16.5 m^3/day. MAR systems with an infiltration rate at the lower end of the estimate would therefore be unsuitable in the southern part of the area, but the low estimates are still higher than our assumed value of 5 m^3/day. This large range underlines the importance of taking the size of the pond (as key rainfall harvesting structure) into consideration during site selection. For MAR sites to have sufficient capacity to function well in the south, it is thus essential to have a large pond or to incorporate additional ways of collecting water, for example, from large roofs.

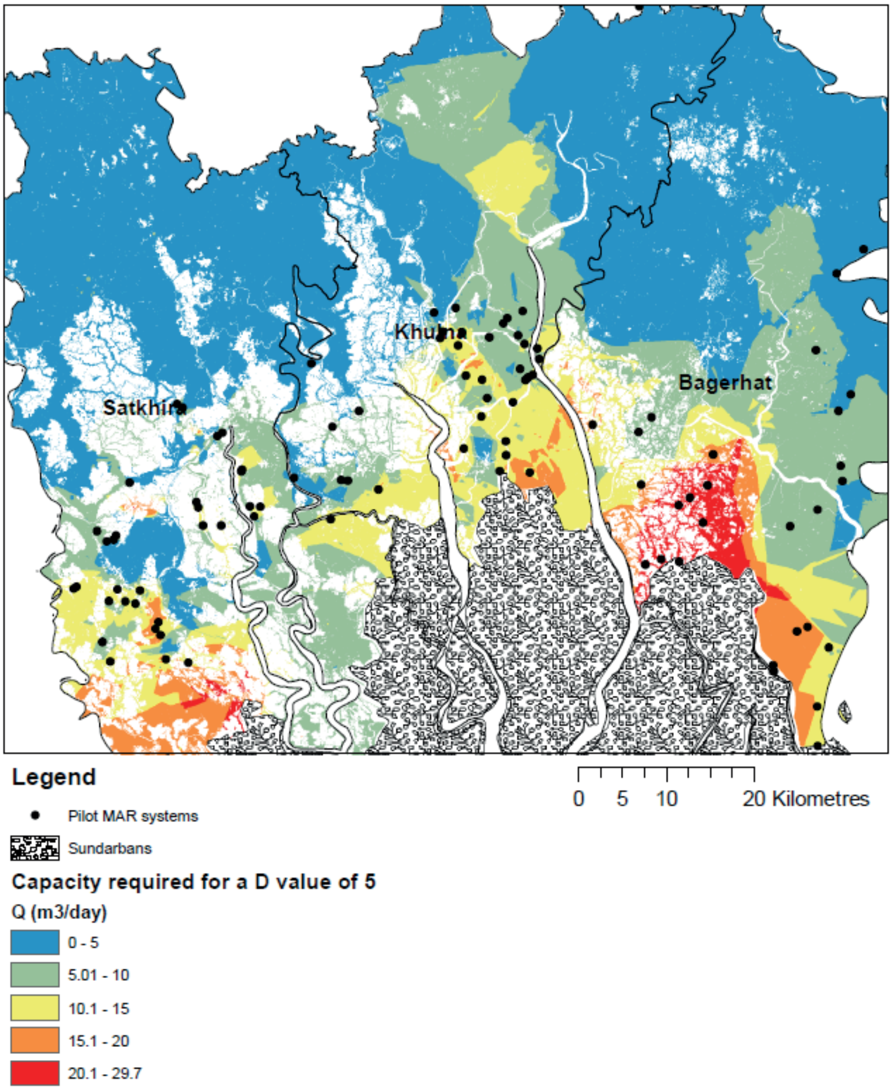


Figure 5.8. Required infiltration (and abstraction rate) per day (capacity MAR system) to achieve a D value of 5, corresponding with a recovery efficiency of approximately 60%.

5.4.4 Reflection on uncertainties

The results were not intended to provide the definite locations where MAR should be applied but rather to assist in developing a strategy for the implementation of MAR throughout the region. They should therefore be seen as providing regional indications for the potential for MAR. Below, uncertainties in the results arising from the methodology, local hydrogeological variation and locally available alternative drinking water options are discussed.

5.4.4.1 Methodological uncertainty

The technical suitability and required infiltration rate are only indicative, as the D value was not specifically developed for the design and operational conditions of these specific MAR systems and required interpretation (section 2.3.2 ii). The effective storage time of the water in the MAR systems might be much shorter than we assumed, which would allow for less time for density-driven flow and an associated higher RE for similar D values. Similarly, our interpretation that the smaller susceptibility to density-driven flow due to infiltration and abstraction at different depths leads to a more efficient system may not hold true for these small MAR sites; Barker et al. (2016), who modelled a pilot MAR system (albeit with different assumptions), found that the design is more efficient when infiltration and abstraction are applied along the entire depth of the aquifer.

5

An additional uncertainty is introduced by the hydraulic conductivity value we used when calculating the D value, namely the median value of 10.9 m/day (σ: 5.68) from 10 pumping tests performed by Acacia (2014b). The hydraulic conductivity of the aquifer has a direct effect on the assessed effect of density-driven flow: when the hydraulic conductivity is twice as low, the D value is also twice as low, when it is twice as high, the D value is twice as high. The effect of hydraulic conductivity is clear when or results are compared with those of Barker et al. (2016), who modelled a pilot MAR system using an aquifer hydraulic conductivity of 0.2 m/day. Their lower hydraulic conductivity affects the D value by a factor of 50. Consequently, their model predicted limited influence of density-driven flow, despite the native groundwater being relatively saline.

5.4.4.2 Local hydrogeological variation

Our results reveal the main patterns at a regional scale but there may be deviations at local scale because of the local hydrogeological variation not captured by the detailed dataset and the effects of smoothing by the Kriging. Even though we used the most exhaustive groundwater quality databases available, the area is known for its large local hydrogeological variation (Ayers et al., 2016; Naus et al., 2019a; Naus et al., 2019b). Any local variation in the groundwater quality or hydrogeology not present in the databases used is missed in the interpolated Kriging maps. As mentioned in the results, the Kriged aquifer thickness predicts the aquifer to be thicker than 9.1 m throughout the region, despite 48 out of 875 borehole logs indicating a thinner aquifer. Similarly, the hydraulic conductivity of the aquifer is likely to have large local variation, suggesting a local K value should be estimated during local site assessment. For the water quality, it is also likely that some of the local variation is smoothed during Kriging. As a consequence, the necessity and technical suitability for MAR on a local scale may be different than inferred from the regional overview.

In addition, the assumption that the lateral regional groundwater flow is negligible might not hold true in some local circumstances. On a local scale, it is possible that lateral flow (e.g. caused by pumping nearby) can affect the functioning of the MAR systems. For these reasons, MAR implementation should always be preceded by assessing local hydrogeological features at the site of interest.

5.4.4.3 Local alternative water options

Another uncertainty concerning the applicability at the local scale involves the assessed drinking water options that determine the necessity for MAR. We considered the main water options in the region (rainwater, pond water, groundwater) but there are some other less used alternative drinking water options, although adequate information on their presence is not available. For example, Reverse Osmosis (RO) systems have been implemented recently in the region. These systems can remove bacteria, salinity and arsenic from water (Islam et al., 2018; Ning 2002). However, the construction costs, maintenance costs, and energy demand of these RO systems are all high and their robustness over longer time periods is not yet known: RO systems in southwestern Bangladesh are reported to not always succeed in lowering the salinity of the water (Islam et al., 2018). Nevertheless, the presence of a well-maintained and operated RO system could be an alternative to MAR. Pipeline systems which distribute good quality groundwater to nearby areas are another alternative drinking water option being increasingly invested in (Hoque et al., 2019). These systems are generally well accepted by the local population (Inauen et al., 2013). A well-functioning piped water supply system could be an alternative safe drinking water option, although installation and maintenance of piped systems is expensive, with costs increasing and quality decreasing with water transported over larger distances; furthermore, the provision of good quality deep groundwater through the pipelines may not be sustainable if the source is fossil groundwater pockets.

5.4.5 Comparison of assessment with pilot MARs

To assess the predictive value of the technical suitability map, it was compared with the observed performance of the pilot MAR sites. As the RE of the pilot MAR systems is not known because usually less is abstracted than has infiltrated, we used the quality of the abstracted water from the pilot MAR systems as an indicator of technical suitability. However, this is not a comparison without complications. A MAR system with an intrinsically low RE could be producing water of sufficient quality because much more water infiltrates than is abstracted. Similarly, a MAR system with a high RE could still at some moment produce water of insufficient quality if its popularity results in abstraction exceeding recharge. Nevertheless, this comparison was made, due to a lack of an alternative. There are 86 pilot MAR sites for which sufficient water quality data was available. The pilot sites are relatively evenly distributed across the first three technical suitability classes (Table 5): 20 in the class with a technical suitability lower than 0.5, 33 in the class with a technical suitability between 0.5 and 0.75, and 27 in the class with a technical suitability between 0.75 and 1. However, there are only 6 pilot MAR sites in the class with a technical suitability of 1.

For the comparison, we first determined whether the MAR sites have ever been capable of producing water with an arsenic concentration and salinity within the Bangladesh drinking water standards. This was the case for 68 of the 86 MAR sites. By comparison with the suitability classes, there is a slight tendency for observed performance to improve concomitantly with technical suitability (rising from 75% to 83%) but the differences are not clearly visible (Table 5). This reveals that the MAR sites have the potential to produce water of sufficient quality even in areas having a low technical suitability. However, this only reveals the initial performance of the MAR systems. The best water quality was typically reached before the onset of the dry season, when net abstraction occurred. Therefore, a second comparison was made with the status of the MAR sites after they had been in operation for a longer time period and after the dry season. Here we determined whether the MAR systems were still producing water of sufficient quality at 20-06-2016 (end of dry season in 2016). The results show that only 30 of the 86 pilot MAR sites still delivered water of sufficient quality, indicating that producing water of sufficient quality over longer time and throughout the dry season is more challenging for the pilot MAR sites. There are no clear differences between the technical suitability classes, as MAR sites in all classes perform worse than before, except the MAR sites with a technical suitability score of 1. The latter score may be an indication of the predictive value of at least the class of best hydrogeological suitability.

The general lack of a clear relationship between the suitability index and the recorded performance of the pilot MAR sites suggests there are other factors that control the quality of the produced pilot MAR water. We recommend a more detailed assessment of the performance of the pilot MAR sites by determining the RE of the different pilot MAR sites. For a proper comparison, sites that have not operated continuously should be excluded.

Table 5.5 The best and worst performances of the pilot MAR sites and their performance after longer/prolonged operation on 20-06-2016 (end of dry season), as compared to the MAR technical suitability classes.

		Best performance		Performance at end of dry season (20-06-2016)	
Technical suitability classes	Number of pilot MAR sites	At least once produced water of sufficient quality	Never produced water of sufficient quality	Sufficient quality on 20-06-2016	Insufficient quality on 20-06-2016
0.5-	20	15 (75%)	5 (25%)	7 (35%)	13 (65%)
0.5-0.75	33	26 (79%)	7 (21%)	10 (30%)	23 (70%)
0.75-1	27	22 (82%)	5 (19%)	10 (37%)	17 (63%)
1	6	5 (83%)	1 (17%)	3 (50%)	3 (50%)
Grand Total	86	68 (79%)	18 (21%)	30 (35%)	56 (65%)

5.5 CONCLUSION

This study aimed to reveal the potential for MAR in a region with high arsenic and saline groundwater by combining an overview of the necessity and technical suitability for MAR. This overview was established using the largest compiled groundwater quality dataset in southwestern Bangladesh to date, containing 3716 salinity measurements and 827 arsenic datapoints. For the technical suitability, the impact of density-driven flow on fresh RE by the MAR systems in brackish groundwater environments was determined and the vulnerability of recovered water to mixing with the native brackish-saline or arsenic-contaminated groundwater was assessed.

The results show that protected Sundarbans and the often-inundated areas (containing aquaculture and (tidal) river floodplains) largely limit areas where MAR systems can be installed. In areas with groundwater that is brackish or has a high arsenic content, the MAR systems are expected to function technically well, but for implementation, it is recommended to evaluate the feasibility of community-run MAR systems. In the saline southern parts of the region, where pond water is likely to be consumed, the community-run MAR systems are more likely to be adopted, but for implementation it is recommended to evaluate the MAR design and to construct MAR systems with a high infiltration rate to limit impacts of density-driven flow and MAR water quality deterioration. The calculated density-driven flow can be translated into a practical guideline needed to achieve the approximate MAR infiltration rate that ensures sufficiently high recovery efficiencies.

Compared to the commonly used expert judgement methods, our approach has advantages: 1. it is transparent and verifiable and 2. it provides practical insight into where MAR is needed and how successful implementation of MAR systems could be facilitated. Therefore, the results are valuable for designing a regional strategy for MAR implementation. The approach may be useful for mapping integrated social-technical MAR potential in other saline deltas worldwide.

Acknowledgements.

This work is part of the Delta-MAR project funded by the Urbanising Deltas of the World (UDW) programme of the Dutch Research Council (NWO-WOTRO) (Grant number: OND1357179). We would like to thank Chelsea Peters, Steven Goodbred and John Ayers from Vanderbilt University for allowing us to use their data on the region. Additionally, we would like to acknowledge the help of all the staff from the Delta-MAR office in Khulna, particularly Abir Delwaruzzaman, for their support during the fieldwork campaigns.

APPENDIX A

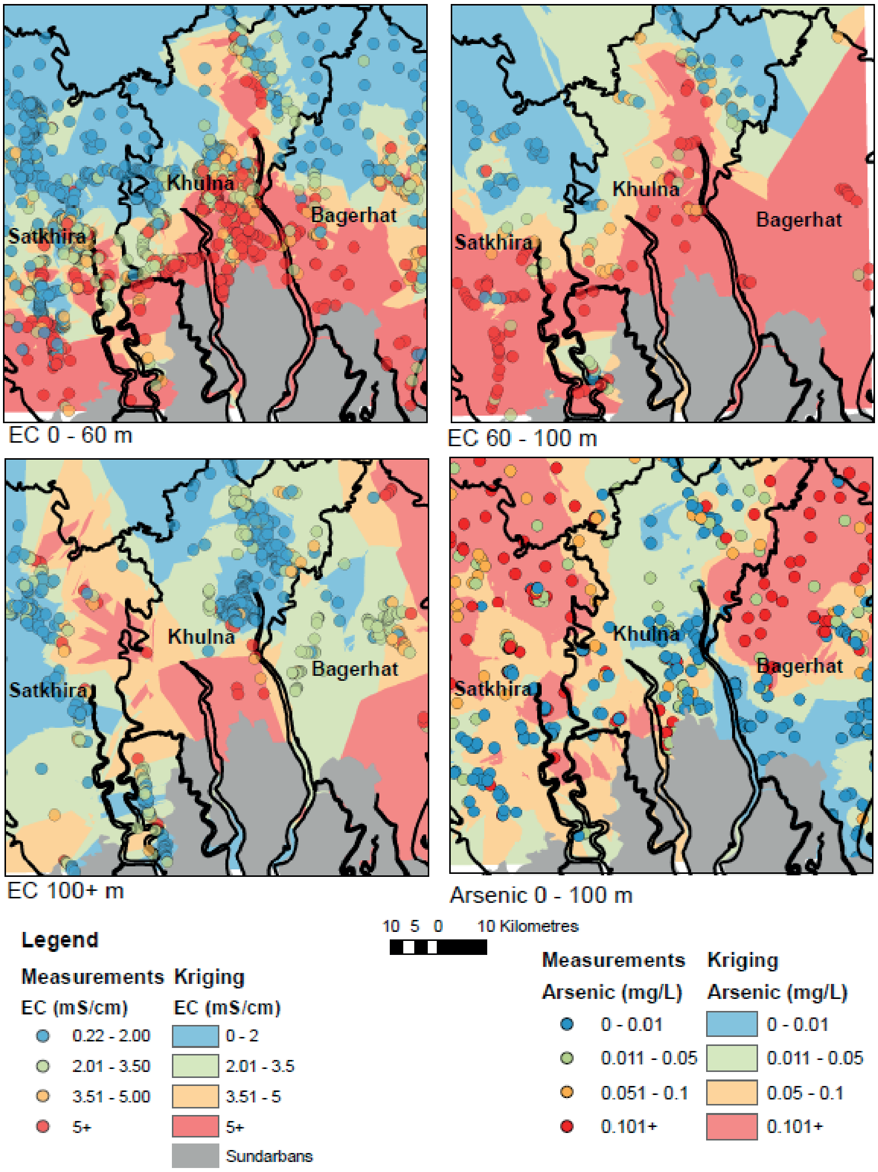


Figure A5.1. Kriging maps showing the EC of the shallow (a), intermediate deep (b), and deep groundwater (c), and the arsenic of the shallow to intermediate deep groundwater (d).

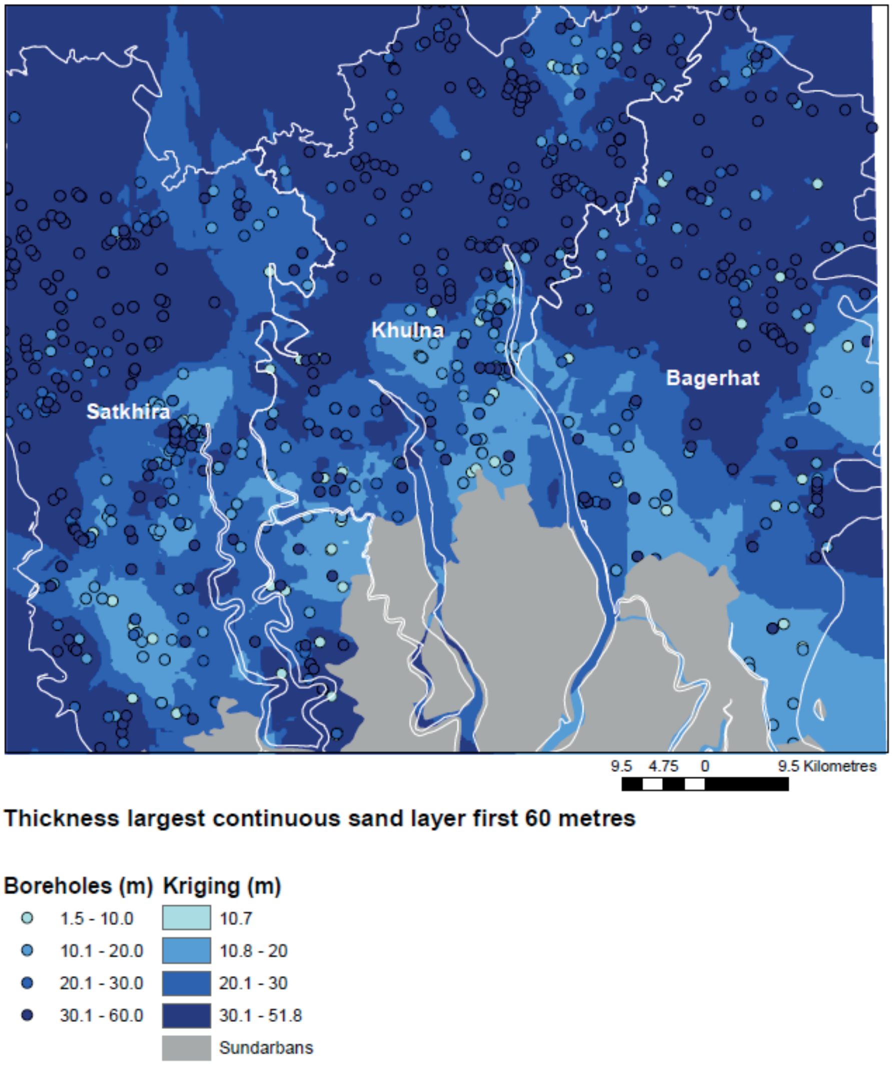


Figure A5.22. Kriging results of the thickness of the aquifer in the first 60 metres.

REFERENCES

Acacia Water, 2014a. Technical Report No. 6. Manual for MAR system implementation in saline areas, Acacia water, Gouda

Acacia Water, 2014b. Technical Report No. 4. Testing and evaluation of MAR system performance, Acacia water, Gouda

Ayers, J.C., Goodbred, S., George, G., Fry, D., Benneyworth, L., Hornberger, G., Roy, K., Karim, M.R., Akter, F., 2016. Sources of salinity and arsenic in groundwater in southwest Bangladesh. Geochem. Trans. 17, 4. https://doi.org/10.1186/s12932-016-0036-6

Bakker, M., 2010. Radial Dupuit interface flow to assess the aquifer storage and recovery potential of saltwater aquifers. Hydrogeol. J. 18, 107–115.https://doi.org/10.1007/s10040-009-0508-1

Bangladesh Water Development Board, 2013. Hydrogeological Study and Mathematical Modelling to Identify Sites for Installation of Observation Well Nests, Selection of Model Boundary, Supervision of Pumping Test, Slug Test, Assessment of Different Hydrogeological Parameters Collection and Conduct Chemical Analysis of Surface Water and Groundwater, Bangladesh Water Development Board, Dhaka.

British Geological Survey (BGS) and Department of Public Health Engineering (DPHE), 2001. Arsenic contamination of groundwater in Bangladesh; BGS Technical Report WC/00/19. British Geological Survey, Keyworth.

Brown, C.J., Weiss, R., Verrastro, R., Schubert, S., 2005. Development of an Aquifer, Storage and Recovery (ASR) site selection suitability index in support of the Comprehensive Everglades Restoration project. J. Environ. Hydrol. 13.

Chowdhury, A., Jha, M.K., Chowdary, V.M., 2009. Delineation of groundwater recharge zones and identification of artificial recharge sites in West Medinipur district, West Bengal, using RS, GIS and MCDM techniques. Environ. Earth Sci. 59, 1209–1222. https://doi.org/10.1007/s12665-009-0110-9

Chowdhury, N.T., 2010. Water management in Bangladesh: an analytical review. Water Policy 12, 32. https://doi.org/10.2166/wp.2009.112

Despins, C., Farahbakhsh, K., Leidl, C., 2009. Assessment of rainwater quality from rainwater harvesting systems in Ontario, Canada. J. Water Supply Res. Technol. – AQUA 58, 117–134. https://doi.org/10.2166/aqua.2009.013

Dillon, P., 2005. Future management of aquifer recharge. Hydrogeol. J. 13, 313–316. https://doi.org/10.1007/s10040-004-0413-6

Dobrowsky, P.H., De Kwaadsteniet, M., Cloete, T.E., Khan, W., 2014. Distribution of Indigenous Bacterial Pathogens and Potential Pathogens Associated with Roof-Harvested Rainwater. Appl. Environ. Microbiol. 80, 2307–2316. https://doi.org/10.1128/aem.04130-13

Fendorf, S., Michael, H.A., Geen, A. Van, 2010. Spatial and Temporal Variations of Groundwater Arsenic in South and Southeast Asia — Supporting Online Material 1123–1128.

Flanagan, S., Johnston, R., Zheng, Y., 2012. Arsenic in tube well water in Bangladesh:

health and economic impacts and implications for arsenic mitigation. Bull. World Health Organ. 90, 839–846. https://doi.org/10.2471/BLT.11.101253

Fofonoff, N.P., Millard, R.C.J., 1983. Algorithms for the computation of fundamental properties of seawater. Unesco Tech. Pap. Mar. Sci., vol. 44, 58. UNESCO, Paris (France): Division of Marine Sciences.

George, G.J., 2013. Characterization of salinity sources in southwestern Bangladesh evaluated through surface water and groundwater geochemical analyses. Vanderbilt University.

Ghayoumian, J., Mohseni Saravi, M., Feiznia, S., Nouri, B., Malekian, A., 2007. Application of GIS techniques to determine areas most suitable for artificial groundwater recharge in a coastal aquifer in southern Iran. J. Asian Earth Sci. 30, 364–374. https://doi.org/10.1016/j.jseaes.2006.11.002

Harvey, C.F., Swartz, C.H., Badruzzaman, A.B.M., Keon-Blute, N., Yu, W., Ali, M.A., Jay, J., Beckie, R., Niedan, V., Brabander, D., Oates, P.M., Ashfaque, K.N., Islam, S., Hemond, H.F., Ahmed, M.F., 2002. Arsenic Mobility and Groundwater Extraction in Bangladesh. Science (80-.). 298, 1602–1606. https://doi.org/10.1126/science.1076978

Hasan M.M., Ahmed K.M., Sultana S., Rahman M.S., Ghosh S.K., Ravenscroft P. (2018) Investigations on Groundwater Buffering in Khulna-Satkhira Coastal Belt Using Managed Aquifer Recharge. In: Mukherjee A. (eds) Groundwater of South Asia. Springer Hydrogeology. Springer, Singapore

Hoque, M.A., Hoque, M.M., Ahmed, K.M., 2007. Declining groundwater level and aquifer dewatering in Dhaka metropolitan area, Bangladesh: Causes and quantification. Hydrogeol. J. 15, 1523–1534. https://doi.org/10.1007/s10040-007-0226-5

Hoque, S.F., Hope, R., Arif, S.T., Akhter, T., Naz, M., Salehin, M., 2019. A social-ecological analysis of drinking water risks in coastal Bangladesh. Sci. Total Environ. 679, 23–34. https://doi.org/10.1016/j.scitotenv.2019.04.359

Horneman, A., van Geen, A., Kent, D.V., Mathe, P.E., Zheng, Y., Dhar, R.K., O'Connell, S., Hoque, M.A., Aziz, Z., Shamsudduha, M., Seddique, A.A., Ahmed, K.M., 2004. Decoupling of As and Fe release to Bangladesh groundwater under reducing conditions. Part I: Evidence from sediment profiles. Geochim. Cosmochim. Acta 68, 3459–3473. https://doi.org/10.1016/j.gca.2004.01.026

Inauen, J., Hossain, M.M., Johnston, R.B., Mosler, H.J., 2013. Acceptance and Use of Eight Arsenic-Safe Drinking Water Options in Bangladesh. PloS One 8. https://doi.org/10.1371/journal.pone.0053640

Islam, M.M., Chou, F.N.F., Kabir, M.R., Liaw, C.H., 2010. Rainwater: A Potential Alternative Source for Scarce Safe Drinking and Arsenic Contaminated Water in Bangladesh. Water Resour. Manag. 24, 3987–4008. https://doi.org/10.1007/s11269-010-9643-7

Islam, M.A., Sakakibara, H., Karim, M.R., Sekine, M., Mahmud, Z.H., 2011. Bacteriological assessment of drinking water supply options in coastal areas of Bangladesh. J. Water Health 9, 415–428. https://doi.org/10.2166/wh.2011.114

Islam, M.A., Akber, M.A., Ghosh, P.K., 2018. Water quality of small-scale desalination plants in southwest coastal Bangladesh. Water Sci. Technol. Water Supply 18, 1606–1616. https://doi.org/10.2166/ws.2017.222

Kabir, M.P., Islam, M.A., Akber, M.A., Ahsan, M.S., 2016. Bacteriological assessment of aquifer storage and recovery (ASR) water in southwest coastal Bangladesh. J. Water Sanit. Hyg. Dev. 6, 231–241. https://doi.org/10.2166/washdev.2016.171

Kallali, H., Anane, M., Jellali, S., Tarhouni, J., 2007. GIS-based multi-criteria analysis for potential wastewater aquifer recharge sites. Desalination 215, 111–119. https://doi.org/10.1016/j.desal.2006.11.016

Khan, A.E., Scheelbeek, P.F.D., Shilpi, A.B., Chan, Q., Mojumder, S.K., Rahman, A., Haines, A., Vineis, P., 2014. Salinity in Drinking Water and the Risk of (Pre)Eclampsia and Gestational Hypertension in Coastal Bangladesh: A Case-Control Study. PLoS One 9, e108715. https://doi.org/10.1371/journal.pone.0108715

Knappett, P.S.K., Escamilla, V., Layton, A., McKay, L.D., Emch, M., Williams, D.E., Huq, R., Alam, J., Farhana, L., Mailloux, B.J., Ferguson, A., Sayler, G.S., Ahmed, K.M., van Geen, A., 2011. Impact of population and latrines on fecal contamination of ponds in rural Bangladesh. Sci. Total Environ. 409, 3174–3182. https://doi.org/10.1016/j.scitotenv.2011.04.043

Kränzlin, I., 2000. Pond management in rural Bangladesh: problems and possibilities in the context of the water supply crisis. Nat. Resour. Forum 24, 211–223. https://doi.org/10.1111/j.1477-8947.2000.tb00945.x

Lowry, C.S., Anderson, M.P., 2006. An assessment of aquifer storage recovery using ground water flow models. Ground Water 44, 661–667. https://doi.org/10.1111/j.1745-6584.2006.00237.x

Maliva, R.G., Guo, W., Missimer, T.M., 2006. Aquifer Storage and Recovery: Recent Hydrogeological Advances and System Performance. Water Environ. Res. 78, 2428–2435. https://doi.org/10.2175/106143006X123102

Mukherjee, A., Fryar, A.E., Thomas, W.A., 2009. Geologic, geomorphic and hydrologic framework and evolution of the Bengal basin, India and Bangladesh. J. Asian Earth Sci. 34, 227–244. https://doi.org/10.1016/j.jseaes.2008.05.011

Naus, F.L., Schot, P., Groen, K., Ahmed, K.M., Griffioen, J., 2019a. Groundwater salinity variation in Upazila Assasuni (southwestern Bangladesh), as steered by surface clay layer thickness, relative elevation and present-day land use. Hydrol. Earth Syst. Sci. 23, 1431–1451. https://doi.org/10.5194/hess-23-1431-2019

Naus, F.L., Schot, P., Ahmed, K.M., Griffioen, J., 2019b. Influence of landscape features on the large variation of shallow groundwater salinity in southwestern Bangladesh. J. Hydrol. X 5. https://doi.org/10.1016/j.hydroa.2019.100043

Naus, F. L., Burer, K., van Laerhoven, F., Griffioen, J., Ahmed, K. M., & Schot, P. (2020). Why Do People Remain Attached to Unsafe Drinking Water Options? Quantitative Evidence from Southwestern Bangladesh. Water, 12(2), 342.

Nickson, R., McArthur, J., Burgess, W., Ahmed, K.M., Ravenscroft, P., Rahmanñ, M., 1998. Arsenic poisoning of Bangladesh groundwater. Nature 395, 338–338. https://doi.org/10.1038/26387

Ning, R.Y., 2002. Arsenic removal by reverse osmosis. Desalination 143, 237–241. https://doi.org/10.1016/S0011-9164(02)00262-X

Peters, C.N., Baroud, H., Hornberger, G.M., 2019. Multicriteria Decision Analysis of Drinking Water Source Selection in Southwestern Bangladesh. J. Water Resour. Plan. Manag. 145, 05019004. https://doi.org/10.1061/(ASCE)WR.1943-5452.0001029

Post, V.E.A., 2012. Electrical Conductivity as a Proxy for Groundwater Density in Coastal Aquifers. Ground Water 50, 785–792. https://doi.org/10.1111/j.1745-6584.2011.00903.x

Rahman, M.A., Rusteberg, B., Gogu, R.C., Lobo Ferreira, J.P., Sauter, M., 2012. A new spatial multi-criteria decision support tool for site selection for implementation of managed aquifer recharge. J. Environ. Manage. 99, 61–75. https://doi.org/10.1016/j.jenvman.2012.01.003

Rahman, A.K.M.M., Ahmed, K.M., Butler, A.P., Hoque, A.M., 2018. Influence of surface geology and micro-scale land use on the shallow subsurface salinity in deltaic coastal areas: a case from southwest Bangladesh. Environ. Earth Sci. 77, 423. https://doi.org/10.1007/s12665-018-7594-0

Ravenscroft, P., Burgess, W.G., Ahmed, K.M., Burren, M., Perrin, J., 2005. Arsenic in groundwater of the Bengal Basin, Bangladesh: Distribution, field relations, and hydrogeological setting. Hydrogeol. J. 13, 727–751. https://doi.org/10.1007/s10040-003-0314-0

Russo, T.A., Fisher, A.T., Lockwood, B.S., 2015. Assessment of managed aquifer recharge site suitability using a GIS and modeling. Groundwater 53, 389–400. https://doi.org/10.1111/gwat.12213

Saraf, A.K., Choudhury, P.R., 1998. Integrated remote sensing and gis for groundwater exploration and identification of artificial recharge sites. Int. J. Remote Sens. 19, 1825–1841. https://doi.org/10.1080/014311698215018

Shahid, S., Hazarika, M.K., 2010. Groundwater drought in the northwestern districts of Bangladesh. Water Resour. Manag. 24, 1989–2006. https://doi.org/10.1007/s11269-009-9534-y

Sharma, B., Amarasinghe, U., Xueliang, C., de Condappa, D., Shah, T., Mukherji, A., Bharati, L., Ambili, G., Qureshi, A., Pant, D., Xenarios, S., Singh, R., Smakhtin, V., 2010. The Indus and the Ganges: River basins under extreme pressure. Water Int. 35, 493–521. https://doi.org/10.1080/02508060.2010.512996

Sprenger, C., Hartog, N., Hernández, M., Vilanova, E., Grützmacher, G., 2017. Inventory of managed aquifer recharge sites in Europe : historical development , current situation and perspectives. https://doi.org/10.1007/s10040-017-1554-8

Stefan, C., Ansems, N., 2018. Web-based global inventory of managed aquifer recharge applications. Sustain. Water Resour. Manag. 4, 153–162. https://doi.org/10.1007/s40899-017-0212-6

Sultana, S., Ahmed, K.M., Mahted-Ul-Alam, S.M., Hasan, M., Tuinhof, A., Ghosh, S.K., Rahman, M.S., Ravenscroft, P., Zheng, Y., 2014. Low-Cost Aquifer Storage and Recovery: Implications for Improving Drinking Water Access for Rural Communities in Coastal Bangladesh. J. Hydrol. Eng. 20. https://doi.org/10.1061/(ASCE)HE

United Nations, 2015. Transforming Our World: The 2030 Agenda for Sustainable Development - A/RES/70/1. https://doi.org/10.1891/9780826190123.ap02

van Halem, D., Olivero, S., de Vet, W.W.J.M., Verberk, J.Q.J.C., Amy, G.L., van Dijk, J.C., 2010. Subsurface iron and arsenic removal for shallow tube well drinking water supply in rural Bangladesh. Water Res. 44, 5761–5769. https://doi.org/10.1016/j.watres.2010.05.049

Ward, J.D., Simmons, C.T., Dillon, P.J., Pavelic, P., 2009. Integrated assessment of lateral flow, density effects and dispersion in aquifer storage and recovery. J. Hydrol. 370, 83–99. https://doi.org/10.1016/j.jhydrol.2009.02.055

Yokota, H., Tanabe, K., Sezaki, M., Akiyoshi, Y., Miyata, T., Kawahara, K., Tsushima, S., Hironaka, H., Takafuji, H., Rahman, M., Ahmad, S.A., Sayed, M.H.S.U., Faruquee, M.H., 2001. Arsenic contamination of ground and pond water and water purification system using pond water in Bangladesh. Eng. Geol. 60, 323–331. https://doi.org/10.1016/S0013-7952(00)00112-5

Zuurbier, K.G., Bakker, M., Zaadnoordijk, W.J., Stuyfzand, P.J., 2013. Identification of potential sites for aquifer storage and recovery (ASR) in coastal areas using ASR performance estimation methods. Hydrogeol. J. 21, 1373–1383. https://doi.org/10.1007/s10040-013-1003-2

Zuurbier, K.G., Zaadnoordijk, W.J., Stuyfzand, P.J., 2014. How multiple partially penetrating wells improve the freshwater recovery of coastal aquifer storage and recovery (ASR) systems: A field and modeling study. J. Hydrol. 509, 430–441. https://doi.org/10.1016/j.jhydrol.2013.11.057

CHAPTER 6

SYNTHESIS

6.1 INTRODUCTION

The research presented in this thesis has focussed on researching the potential that MAR systems could be successful from a technical and a social point-of-view in the complex socio-hydrogeological settings of southwestern Bangladesh. A large part of the study focussed on elucidating the previously largely unknown hydrogeological situation in the region, in specific related to the variation in groundwater salinity. The hydrogeological situation determines whether an area will facilitate a technically well-performing MAR system, and the hydrogeological situation is also related to the necessity for MAR, as it determines whether local groundwater can safely be used for drinking. To elucidate the social potential for MAR, the research in this thesis additionally focussed on the attachment of people to their unsafe water options in the region, as this influences their aptitude to accept MAR. Lastly, regional differences in necessity and technical suitability for MAR were combined, to elucidate potential for MAR and to assess and improve MAR implementation in the region.

In specific, the previous chapters enabled to:

- Develop a conceptual model of the spatio-temporal groundwater salinity evolution through the Holocene to understand the spatial groundwater salinity variation as a function of the controlling palaeo and present-day processes;
- Understand and link these controlling processes to surface characteristics and groundwater salinity at a regional scale;
- Assess to what extent and why users are attached to their current, unsafe water options;
- Determine the regional potential for MAR based on regional differences in necessity and technical suitability

Below, the individual chapters are summarized. Then, implications of the findings for the implementation of MAR systems in southwestern Bangladesh are discussed. Next, the implications of the findings towards other coastal regions are generalized. Last, recommendations for further research are provided.

6.2 SUMMARY OF THE MAIN FINDINGS

6.2.1 Groundwater salinity variation at a local scale

Chapter 2 presents the results of the local-scale study on the hydrogeological and groundwater salinity variations. The aim of the study was to determine the palaeo and present-day hydrological processes and the geoscientific controls that determine the large variation in groundwater salinity at a local scale. The study was based on fieldwork to collect high-density hydrological and lithological data along a 5 km transect. Groundwater freshening and salinization patterns were deduced using PHREEQC simulations on groundwater transport coupled with cation exchange, and isotope data were used to derive relevant hydrological processes and recharging water types. Subsequently, the collected data were linked to a geological reconstruction based on the literature in order to infer the spatio-temporal evolution of the groundwater salinity in the study area. The results of the study show how the large variation in salinity observed can be explained by a variety of palaeo and present-day hydrological processes, with the thickness of the Holocene clayey top layer steering the relative importance of palaeo versus present-day hydrogeological conditions. The groundwater in aquifers under thick surface clay layers is controlled by the palaeohydrological conditions prevailing when the aquifers were buried. Groundwater in aquifers under thin surface clay layers is affected by present-day processes, which vary depending on present-day surface elevation. Recharge by rain and rainfed ponds is present in slightly higher-lying areas and, therefore, fresh groundwater is located at shallow depths. In contrast, the lower-lying areas with a thin surface clay layer have brackish–saline groundwater at shallow depth because of flooding by marine-influenced water, subsequent infiltration and aquifer intrusion. Recently, infiltration from aquaculture ponds in areas having a thin clayey top layer increased the salinity in the underlying shallow aquifers. The relative elevation and land use can be used as a first estimate in areas with a thin surface clay layer, while knowledge of palaeohydrogeological conditions is needed in areas with a thick surface clay layer.

6.2.2 Groundwater salinity variation on a regional scale

Chapter 3 presents the results of a regional study on the variation in salinity of shallow (< 60 m) groundwater. The aim of the study was to assess the regional salinity variation as a function of landscape features and associated hydrological processes. The spatial variation in groundwater salinity was assessed using 442 EC measurements from previous studies and 1998 newly collected EC measurements, with EC being used as an indicator for Cl concentration in groundwater. Groundwater EC values were correlated with well location data (latitude, longitude and depth of the filter) and landscape characteristic data (elevation, soil type, land use and surface clay thickness). Additionally, a geomorphological analysis of landscape features was performed to infer associated hydrological processes. Wide fluvial zones were interpreted to be remnants of sandy deposits in large paleo channels which allow or allowed freshwater recharge, resulting in groundwater that is mostly (75%) fresh. Narrow fluvial zones, tidal fluvial zones, and fluvial zones next to tidal rivers are more susceptible to lateral inflow of saline water or saline water recharge by occasional tidal flooding, and only contain some shallow fresh groundwater in high-lying zones as formed by freshwater recharge. Tidal flat or tidal fringe zones are influenced by saline water recharge, and only contain some fresh groundwater formed under palaeo conditions. This study is the first to demonstrate the relation between landscape features, hydrological processes and regional groundwater salinity throughout southwestern Bangladesh. The main lines of this approach may be applicable in other coastal areas with available data on spatial landscape features, enabling a first prediction of groundwater salinity variation.

6.2.3 Attachment to unsafe water options

Chapter 4 presents the results of an interview study on the attachment of the local people to their current water options. The aim of the study was to determine why, as well as the extent to which, people are expected to remain attached to using these unsafe pond and shallow tube wells, compared to the following four safer drinking water options: deep tubewells, pond sand filters, vendor water, and rainwater harvesting. Through 262 surveys, was explored whether five explanatory factors (risk, attitude, norms, reliability, and habit) pose barriers to switching from unsafe to safe drinking water options or whether they could act as facilitators of such a switch.

The results show that users’ attachment to pond water is generally low. Factors facilitating a switch towards safer alternatives are perceived risk and attitude, whereas norms are a barrier for such a switch. For STW, reliability and habit are barriers preventing a switch to safer options, and there are no clear facilitators for such a switch. STW users are more attached to their source than pond users, and, therefore, safer drinking water options have less potential to replace STWs than to replace pond water. The safe alternatives (deep tubewell, rainwater harvesting, pond sand filter, and vendor water) score significantly better than pond water and are estimated to have the potential to be adopted by pond water users. Deep tubewell, rainwater harvesting, and pond sand filter also score better than shallow tubewells and may also have the potential to replace them. These findings may be used to optimise implementation strategies for MAR.

6.2.4 Regional MAR potential based on necessity and technical suitability

Chapter 5 presents the research on the potential for MAR throughout southwestern Bangladesh. The aim of the study was to assess differences in potential for MAR throughout the region using an approach that combines 1) the necessity of local communities for MAR based on the groundwater quality present, and 2) the technical MAR suitability based on constraints for MAR and the expected effect of density-driven flow and vulnerability to mixing. During these two assessments, we used land cover data, a lithological database, and a groundwater quality database. We mapped the existing groundwater quality using the largest compiled groundwater quality dataset in southwestern Bangladesh to date (salinity, n=3716; arsenic, n= 827).

The results show the often-inundated areas (containing aquaculture and (tidal) river floodplains) largely limit areas where MAR systems can be installed. In the rest of the region, there is a general mismatch between necessity and technical suitability. In some northern parts of the region, necessity is low because groundwater of good quality is present, reducing the potential for MAR, despite the high technical suitability. In other northern parts of the region, unsafe arsenic or brackish groundwater is likely to be used for drinking water. There, MAR is a technical suitable and safer drinking water option. In southern parts of the region, groundwater is saline at many places, causing people to resort to consumption of bacterially unsafe pond water. Especially in these parts there is a high necessity for MAR as safe drinking water option. However, the high groundwater salinity calls for careful evaluation of MAR design and calls for implementing MAR systems with a high infiltration rate to limit impacts of density driven flow and MAR water quality deterioration. The calculated density driven flow can be translated into a practical guideline for the approximate MAR infiltration rate to achieve sufficiently high recovery efficiencies. The developed approach may be useful for mapping integrated social-technical MAR potential in other saline deltas world-wide.

6.3 MAR IMPLEMENTATION IN SOUTHWESTERN BANGLADESH

In this section, the findings are evaluated with a focus on implementation of MAR systems. In specific, implications of the findings are discussed for the following three topics: 1) local MAR site-selection; 2) acceptance of MAR; and 3) other factors to consider for MAR implementation. The implications are useful for government organizations, NGOs and other organizations that want to consider implementation of MAR systems to improve the drinking water situation in southwestern Bangladesh. Additionally, the implications may be useful for the implementation of MAR systems for other purposes, such as to provide irrigation water. While these implications are tailored to the situation in the study area, they may also provide useful lessons for the implementation of MAR systems in other areas in the world.

6.3.1 Local MAR site-selection

Chapter 5 provides a regional overview of the potential for MAR. However, local site-selection is also required for MAR implementation. Therefore, the local necessity and hydrogeological conditions should be researched. For the local site-selection, it is recommendable to apply the findings of chapters 2 and 3, as they provide a system-based, conceptual understanding of the hydrogeological situation, groundwater salinity, and steering palaeo and present-day processes.

Firstly, the system-based understanding presented in chapters 2 and 3 can be used to estimate the native groundwater salinity. The MAR systems are expected to perform well in locations where the salinity of the native groundwater is low, so a detailed approximation of the groundwater salinity is important during local MAR site-selection. For this, the flowchart presented in chapter 3 (Figure 3.7) is an especially beneficial practical tool. After this system-based estimation, the local conditions should always be checked by measurements to account for the large local variability in hydrogeological conditions.

Secondly, the system-based understanding from chapters 2 and 3 can also be used to assess whether relevant present-day hydrological processes may influence the MAR systems during operation. In areas with a thick clay layer, the groundwater was found to be isolated, which means that MAR systems in these locations will likely not be subject to lateral groundwater flow or pollution from the surface. In contrast, groundwater under thin clay layers was found to be influenced by present-day processes, where it is, therefore, important to assess possible influences of lateral flow and potential pollution pathways. Possible relevant present-day infiltration processes identified in chapters 2 and 3 are freshwater recharge in high-lying areas, and saline water infiltration from tidal rivers, saline aquaculture ponds or tidal flooding in the low-lying areas. Although infiltration rates and groundwater flow paths in this thesis have not been researched in detail, a first attempt of elucidating the groundwater flow pathways and rates along the transect in Assasuni is provided in the master thesis of Smidt (2018). Groundwater flow was found to be driven by freshwater recharge at the higher areas, with drainage at the low points in the landscape, i.e., the tidal rivers located at a lower elevation than the polders, and by groundwater pumping for households, irrigation and industry. Additionally, it was found that in the dry season, groundwater flow direction reverses from the inundated areas with aquaculture towards the high-lying village, probably due to a combination of extensive pumping and the fact that the areas with aquaculture remain inundated. These results reveal that MAR systems may be influenced by lateral flow when placed close to the edge of high-lying areas, close to drainage points in the landscape, or close to areas with extensive groundwater extraction for agriculture or industry, e.g. brick factories (Smidt, 2018).

6.3.2 Acceptance of MAR

As discussed in chapter 4, the adoption of safe alternative water sources is not guaranteed even when they would be available and affordable. The findings have implications for the implementation strategy of MAR systems. The results show that pond users are more likely to switch away from their water option compared to STW users. A switch away from ponds is facilitated by users' attitude towards pond water and their perception of associated risks, but obstructed by community norms. A switch from STWs is barriered by the reliability and habit of using STW water, while none of the factors facilitate such a switch. Here will be discussed how MAR systems may be implemented to be more likely accepted as a source for drinking water in favour of ponds and STWs.

Firstly, MAR systems should perform well. A technically well-functioning MAR system is likely to produce water with a good palatability, i.e. with a low iron and salinity concentration. Additionally, reliable MAR systems will ensure that people will trust the water option, count on the MAR systems, and continue using them, facilitating long-term adoption of MAR.

Secondly, the distribution strategy of the water from the MAR systems could be adjusted to improve the attitude of the community towards the MAR system. The current MAR systems were placed in specific locations, where the users will have to walk to collect the water. It is, therefore, likely that the collection time for MAR systems will be similar to that for the protected ponds designated for drinking water purposes but much longer than that for STWs. When replacing ponds, this is not likely to be a problem for the acceptance, but this may inhibit a switch away from STWs which are available close to their homes. The collection time for MAR water could be reduced by applying a distribution method, such as home delivery or piped systems. Such a distribution strategy may also help develop a habit of the users, as it is relatively easy to keep using MAR water when it is delivered close to the household. The financial costs for such systems are, however, higher, which may exclude a part of the people.

Thirdly, the community should be informed and become involved during MAR implementation (Hassan et al., 2020). By involving the community, there is an increased possibility to inform the local communities of the health benefits of switching to MAR water compared to unsafe water options before, during and after implementation. As a consequence, MAR systems may be established as a healthy option compared to the unsafe water options present in the local communities, leading to the risk perception and norms of the community to be positive towards MAR systems. Additionally, involvement of the local communities enables the possibility to directly take the wishes and feedback of the local communities into account. By enhancing the service that the MAR systems provide based on wishes and feedback, the attitude of the people towards the MAR systems can be improved.

Lastly, MAR water should be priced competitively. The switch from ponds to MAR systems is not expected to be obstructed when the price of MAR water matches the price that pond water users pay for disinfection of the water. For STW users, the costs are, however, expected to be a barrier for a switch, unless the MAR water or the service of MAR systems is perceived significantly more positively.

6.3.3 Other factors

Besides the necessity and technical suitability for MAR as assessed in chapter 5, there are also other relevant factors that control the potential for MAR during implementation. Here, the following other relevant factors will be discussed: a) the source water for the MAR system; b) clogging; and c) the MAR scale and servable community size.

a) Water source

For MAR systems to function well, they require a large enough quantity of source water. In southwestern Bangladesh, there is a limited number of protected ponds with good quality freshwater. As such, suitable water sources could be a bottleneck during the implementation of MAR systems and the construction of new ponds that can collect enough water should be evaluated.

Water can be collected by capturing rainwater, either directly in the ponds or by collecting and redirecting rain of the surrounding areas towards the pond. The size of the surface area that is needed to collect enough rainwater is based on the amount of rain and the approximate target MAR capacity. When assuming a target MAR recharge rate of 10 m^3 per day, which is expected to be sufficient in a large part of the region, the size of the pond can be evaluated based on the amount of net precipitation. By approximating the net precipitation rate at 2000 mm per year, a recharge rate of 10 m^3 per day would correspond to a minimum surface area of 900 m^2, i.e., an area of about 30 x 30 m, to collect enough rainwater. When aiming to achieve a target recharge rate around 30 m^3 per day, recommendable to combat density driven flow in the saline south part of the region, a surface area of 2700 m^2 or 52 x 52 m would be required. Note that this is an estimate of the amount of water to be collected, as evaporation may be underestimated and a larger turbidity in the dry season can cause the remaining pond water to be unsuitable for recharge.

When new ponds are constructed, or existing ponds are excavated deeper, the water will come into contact with clay that has not been exposed to contact with freshwater. Possibly, the water quality in the pond may deteriorate by diffusion or leaching of soil moisture or water into the pond from this newly-exposed clay, as shown by the 91 water samples of the clayey top layer throughout the region with an approximate depth between 1 and 3 meters as taken during the Master thesis research of Essink (2018) using the open auger boring method (De Goffau et al., 2012). Figure 6.1 shows the chloride concentrations of pore water samples from the surface clay layer throughout the region as classified according to land use. The chloride concentration of the soil water varies, with a median value around 2000 mg/L in agricultural land or in treed villages, and a median value of around 4000 mg/L in aquaculture areas. These chloride concentrations indicate that water from the clayey top layer can deteriorate the pond water that usually has a chloride concentration far below 200 mg/L, although this strongly depends on whether and the amount of water that leaches out of the clay in comparison to the amount of rainwater in the pond.

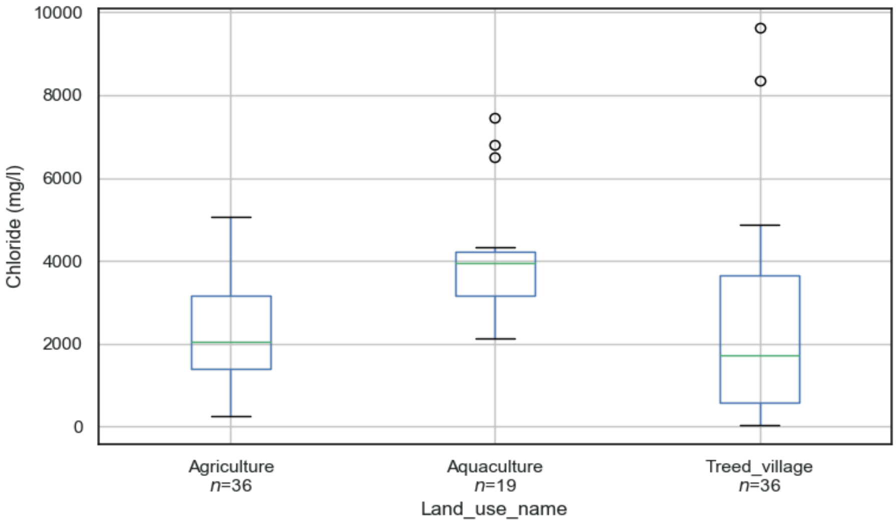


Figure 6.1. Chloride concentration (mg/L) of soil water samples throughout the region, as classified according to the land use.

b) Clogging

During operation, clogging of the wells can decrease the recharge and abstraction rates, which would lead to decreased performance and efficiency of the MAR systems. It is, therefore, important that the MAR systems aim to keep the clogging at a minimal. Firstly, the MAR systems will be susceptible to mechanical clogging when the injection water contains suspended matter. Therefore, possible sources of turbidity should be identified and avoided during the collection and redirection of water into a pond, and the sand filter before the injection well should be properly designed and cleaned to reduce the turbidity of the pond water. An additional benefit of this filtering is that the organic matter in the water is lowered, decreasing the risk of additional reduction causing arsenic mobilization. Secondly, the MAR systems are expected to be susceptible to chemical clogging in the form of iron precipitation, following the introduction of oxygen during injection. For the water quality and taste, the precipitation of iron can be beneficial, as it leads to low iron concentrations and, when arsenic sorbs to the newly formed iron hydroxides, to low arsenic concentrations in the abstracted water (Appelo and de Vet, 2003; Chui et al., 2010; van Halem et al., 2010). Therefore, it can be beneficial to design the MAR system with an overcapacity, so reduction of infiltration and/or abstraction due to the beneficial subsurface iron precipitation will not lead to under capacity. However, if the clogging due to the iron precipitation would be too large, the MAR can be rehabilitated by push-and-pulling acid into the wells to dissolve and remove the precipitated iron oxides.

c) *Scale and community size*

Implementation of larger scale MAR systems has various benefits compared to implementation of smaller MAR systems. As discussed in chapter 5, larger MAR systems are expected to function more efficient, as there will be less contact area between the injected and native groundwater relative to the injected volume of water, and density driven flow will have relatively less influence. Additionally, a larger capacity can enable the MAR water to be designed to have a longer passage through the aquifer, which would lead to a larger attenuation of bacterial contamination from the recharged water.

The scale of the MAR systems has implications for the size of the community that can be provided with MAR water. The servable community size varies according to the scale of the MAR system. According to the surveys in chapter 4, people in southwestern Bangladesh use between 2.5 and 4.5 litre of drinking water per day. Accounting for occasional other uses of the MAR water, for example for washing and cooking, a broad estimate is that MAR systems will have to provide at least 6 litres of water per person per day. When assuming a MAR system can achieve an approximate recovery efficiency of 60%, 10 L person per day is required to be recharged. Chapter 5 provides some approximations of the recharge and abstraction rate required for efficient MAR systems throughout southwestern Bangladesh. These approximations can be used to estimate the ideal servable community size throughout the region. In the northern part of the region having fresh groundwater, relative small-scale MAR systems would already be able to reach a recovery efficiency of 60%, i.e. systems with a recharge rate of 5 m^3 per day (Chapter 5). Such small-scale systems correspond to a servable community of 500 users. Towards the saline southern part of the region, a larger recharge rate is required for a recovery efficiency of 60%, namely up to 30 m^3 per day (Chapter 5). Such larger MAR systems would correspond to a servable community of 3000 users. When communities are smaller than these minima, the MAR systems are either expected to be inefficient, or to have an overcapacity.

To maintain efficient MAR systems while preventing overcapacity in the southern part of the region, these larger MAR systems can be coupled with distribution systems to connect multiple communities or a larger community to one MAR system. Such large MAR systems would make it possible to invest more time and money in the MAR system, which would enable more thorough and stricter site-selection, more fine-tuned design, and more intensive operation and monitoring. These increased efforts are expected to lead to a larger chance of well-performing MAR systems.

6.4 RELEVANCE OF FINDINGS FOR OTHER DELTAS/COASTAL REGIONS

The research in this thesis focusses on assessing the MAR potential in a coastal region with salinity and water quality problems. Zooming out to a global perspective, there are many coastal regions where MAR may be a useful option to overcome current or to prevent future water shortages. The findings and methodology of the research may be relevant for other coastal regions where groundwater contains salinity or faces other water quality issues (Figure 6.2; Van Weert et al., 2009). In this section, I will discuss implications of the findings of this thesis for other saline or brackish coastal regions along three parts of the research: 1. hydrogeology and groundwater salinity; 2. attachment to current water options; and 3. MAR potential mapping.

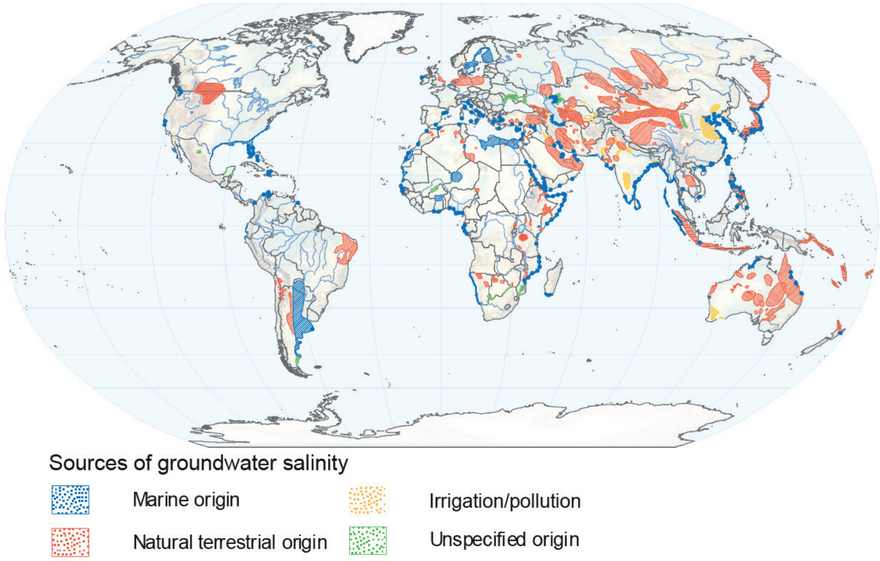


Figure 6.2. Areas in the world with saline groundwater. Figure adjusted from Van Weert et al. (2009).

6.4.1 Hydrogeology and groundwater salinity

In many other coastal regions, groundwater salinity is a problem (Figure 6.2) (Van Weert et al., 2009). While there are coastal regions where the variation in groundwater salinity is largely understood, there are also coastal regions where the variation is not fully understood because of only limited available data (e.g. Mulligan et al., 2011). The lessons and applied methodological framework from this thesis can be valuable when trying to understand and predict the groundwater salinity variation in such regions. Firstly, this thesis shows that it is important to understand the relevant hydrogeological processes and history to understand and predict the groundwater salinity. Especially, the occurrence of regression or transgression periods is important, as they steer the salinity in the deeper, more isolated parts of the subsurface. Many coastal regions are likely to have been influenced by regression and transgression events, but research has not fully elucidated the influence in all coastal areas. Examples of other coastal regions where regression or transgression periods have been described to be relevant are the Red River delta (Tran et al., 2012), the Mekong delta (Wagner et al., 2012), Suriname (Groen et al., 2000), Mozambique (Nogueira et al., 2019), the Pearl River Delta in China (Wang and Jiao, 2012), and the Netherlands (e.g. Oude Essink et al., 2010; Post et al., 2003; Stuyfzand, 1993; Delsman et al., 2014).

Secondly, this thesis shows a methodological framework of linking hydrogeological processes and groundwater salinity to surface characteristics. This framework is mostly relevant in coastal regions where the current salinity variation is not fully understood and where there is a large variation in freshening and salinization processes, either because of variation in lithology or because of a varying elevation close to mean sea level. Examples of other coastal regions where groundwater salinity variation has been shown to be steered by varying processes occurring at different surface elevations, are The Netherlands (Stuyfzand, 1993; Goes et al., 2009; de Louw et al., 2011), Belgium (Walraeven et al., 2007) and Spain (Fernández et al., 2010). It should be noted that processes that are not relevant in southwestern Bangladesh may be important for the groundwater salinity in other coastal areas. For example, human-induced processes involving pollution from the surface, such as road salt (Williams et al., 2000), abstraction-induced saline water upconing or intrusion (Schmorak and Mercado, 1969), such as in the Rhône delta (de Montety et al., 2008), or saline water seepage in polders and other areas below mean sea level, such as in the Netherlands (de Louw et al., 2011; Oude Essink et al., 2010). Saline water seepage is expected to increase in the future due to a combination of land subsidence and sea-level rise (van der Meij & Minnema, 1999).

6.4.2 Attachment to current water options

The findings of the research on the attachment to current water options show that the attachment largely varies between available water options. In regions where people have a choice between multiple available water options, the likeliness of new technologies being successfully implemented expectedly varies. This is most often the case in developing countries, where the supply of drinking water is not centrally managed. Nevertheless, the results are only limitedly applicable in these other regions, because there may be different water options available, and the perception of water options can vary largely between regions, because of differences in cultural and socio-economical situations.

In regions where water supply is centrally managed, implementation of MAR can be directly connected to existing water supply infrastructure. Therefore, the change in technology does not call for a behavioural change regarding water collection, making it is less relevant whether people would accept a new water source or technology.

6.4.3 Assessing MAR potential in other coastal regions

The overall results of the study on the regional MAR potential (chapter 5) can provide lessons for other coastal areas. In other coastal regions, a mismatch between necessity and technical suitability may also be expected. However, it should be noted that MAR potential largely depend on local conditions in other coastal areas, which may call for adjustments when applying the methodology of MAR potential mapping in other regions.

Firstly, the natural groundwater flow may not be negligible in other coastal regions. In southwestern Bangladesh, the groundwater gradient is expected to be low based on a low surface gradient. However, when comparing the slope of deltas as a proxy for the natural groundwater flow velocity that can be expected, there are several deltas with a slope several orders of magnitude higher than the Brahmaputra-Ganges delta (Syvitski and Saito, 2007). The slope of the Brahmaputra-Ganges delta is one of the lowest, with a regional gradient of 0.00009, while the regional gradient of other deltas is much higher, for example the Waipaoa delta (New Zealand) with 0.00466, and the Orange (South Africa), Eel (California) and Huanghe (China) delta with 0.00100. While these regional gradients of the surface are not necessarily indicative for the groundwater head gradient on a local scale, they do reveal that lateral flow is expected to be higher and more relevant in other deltas. As stated in section 6.3.1, groundwater flow may be relevant locally in southwestern Bangladesh, for example where there is a larger surface gradient, close to drainage points of the landscape, or close to abstraction wells. In many other regions, it is likely that groundwater flow will also be induced by anthropogenic activities, such as groundwater extraction or drainage of polders. These abstractions can lead to larger groundwater flow rates and, therefore, higher lateral flow rates at MAR systems.

Secondly, the technical suitability mapping may need to be adjusted when the lithology is different in other coastal regions. In southwestern Bangladesh, the aquifers are confined and consist of very fine to fine sand, causing the flow systems to generally be inert and isolated from pollution from the surface. However, where the conductivity of the soil and aquifer is higher, groundwater flow systems are generally more dynamic and more easily influenced by lateral flow and density driven flow, and by nearby surface pollution sources. In such regions, the suitability mapping should more actively take nearby pollution sources into account. Additionally, it is possible to apply MAR systems that utilize infiltration basins instead of injection wells in regions with a higher soil conductivity, for which a high surface infiltration rate is much more important (Van Breukelen et al., 1998; van Geelen et al., 2017).

6.5 SUGGESTIONS FOR FURTHER RESEARCH IN SOUTHWESTERN BANGLADESH

The research presented in this thesis focussed on southwestern Bangladesh. As the drinking water provision in this region is probably one of the most sensitive areas to future stresses, such as the effects of climate change and an increasing and urbanizing population, more research is expected to be needed on this topic. Here, suggestions are given for future research topics regarding the hydrogeological situation, the drinking water situation, and the application of MAR in southwestern Bangladesh.

Research topic 1: Regional hydrogeological model

This thesis provides a first overview of the regional groundwater salinity variation, and sheds light on the lithological complexity of a small study area. However, there is no clear conceptual model that explains the large lithological variation on a regional scale. Therefore, I recommend future research to focus on developing a more detailed overview of the lithology throughout the region, and the link between the lithology and the salinity on a regional scale. As the collection of borehole data throughout the region is labour intensive and costly, a possible method could involve the application of airborne geophysics. This would make it possible to map the large shallow variation in lithology and groundwater salinity in detail. Because of interference between lithology and groundwater salinity, it is important to calibrate the results in an area where the hydrogeological situation is known and understood. The small-scale study area discussed in chapter 2 would be an ideal calibration location: in this area, the large variation in salinity and lithological conditions is mapped in detail. Aside from applying mapping methods for the shallow variation, the salinity variation in the deeper groundwater is still poorly understood and it is recommendable to study this in more detail.

Research topic 2: Groundwater flow study

Currently, there are many uncertainties about the groundwater flow in the region. In specific, there are many unknowns related to the water balance of the aquifers and the consequential future effects on the groundwater quality. Firstly, the effect of infiltration and drainage points in the region has not been researched in detail. The influence of infiltration and drainage features is complex, as the influence largely depends on the lithological situation, and infiltration and exfiltration rates through the thick clay layers are largely unresearched. Additionally, the influence of human activities on the groundwater flow is unknown. As such, the abstraction of groundwater has not been researched in detail. It is currently unknown how much water is being abstracted, how sustainable the abstraction is, and whether and where this abstraction induces additional infiltration. Lastly, it is unknown what influence aquaculture has or will have on the groundwater. In chapter 2, the aquaculture was linked to possible infiltration of saline water into the subsurface. However, this link is not well made at the regional scale. On geological time scales, aquaculture is a relative new phenomenon, so it may take some time before the influence of the aquaculture is noticeable. Therefore, it is also important to assess to what extent the aquaculture will have additional influences in the future.

Research topic 3: Different designs of MAR

The efficiency of MAR can be increased by optimizing the design of MAR systems depending on the local conditions. MAR systems in relative fresh groundwater environments would call for a different design than MAR systems in relatively saline groundwater environments. A model study could be applied to test how the MAR design can be optimized under various conditions. This model study should focus on simulating hydrogeological scenarios described to be present in the region. By researching how various design choices of MAR systems should be changed according to the local conditions, MAR systems with a higher recovery efficiency can be developed and placed in the region. Examples of changes in designs include the configuration of the injection and abstraction wells according to the groundwater salinity and lateral groundwater head gradient, or how the design of MAR systems should be changed to prevent potential pollution sources to affect the abstracted water.

Research topic 4: Testing acceptance of MAR systems

In this thesis, this thesis researched the factors that control the attachment to unsafe water options. While this gives an indication for how likely it is that MAR systems would be accepted and how MAR can become more accepted, additional research on the controlling factors for the acceptance of MAR systems may reveal more convincingly how the acceptance of MAR systems could be enhanced. During this research, several scenarios of MAR implementation can be tested, such as different distribution strategies, including collection, delivery or piped systems, and different financial constructs, such as paying for the amount of water used, or paying a constant fee per month for the service.

REFERENCES

Ali, S., Thakur, S.K., Sarkar, A., Shekhar, S., 2016. Worldwide contamination of water by fluoride. Environ. Chem. Lett. 14, 291–315. https://doi.org/10.1007/s10311-016-0563-5

Appelo, C. A. J., & de Vet, W. W. J. M. (2003). Modeling in situ iron removal from groundwater with trace elements such as As. Arsenic in Ground Water, 381–401. https://doi.org/10.1007/0-306-47956-7_14

Auerbach, L.W., Goodbred Jr, S.L., Mondal, D.R., Wilson, C. a., Ahmed, K.R., Roy, K., Steckler, M.S., Small, C., Gilligan, J.M., Ackerly, B. a., 2015. Flood risk of natural and embanked landscapes on the Ganges–Brahmaputra tidal delta plain. Nat. Clim. Chang. 5, 153–157. https://doi.org/10.1038/nclimate2472

Cui, Y., Du, X., Weng, L., & Van Riemsdijk, W. H. (2010). Assessment of in situ immobilization of lead (Pb) and arsenic (As) in contaminated soils with phosphate and iron: Solubility and bioaccessibility. Water, Air, and Soil Pollution, 213(1–4), 95–104. https://doi.org/10.1007/s11270-010-0370-8

Delsman, J.R., Hu-A-Ng, K.R.M., Vos, P.C., De Louw, P.G.B., Oude Essink, G.H.P., Stuyfzand, P.J., Bierkens, M.F.P., 2014. Paleo-modeling of coastal saltwater intrusion during the Holocene: An application to the Netherlands. Hydrol. Earth Syst. Sci. 18, 3891–3905. https://doi.org/10.5194/hess-18-3891-2014

De Goffau, A., Van Leeuwen, T. C., Van den Ham, A., Doornewaard, G. J., and Fraters, B.: Minerals Policy Monitoring Programme Report 2007–2010, Methods and Procedures, RIVM Report 680717018, National Institute for Public Health and the Environment, Bilthoven, the Netherlands, 2012

de Louw, P.G.B., Eeman, S., Siemon, B., Voortman, B.R., Gunnink, J., van Baaren, E.S., Oude Essink, G.H.P., 2011. Shallow rainwater lenses in deltaic areas with saline seepage. Hydrol. Earth Syst. Sci. 15, 3659–3678. https://doi.org/10.5194/hess-15-3659-2011

de Montety, V., Radakovitch, O., Vallet-Coulomb, C., Blavoux, B., Hermitte, D., Valles, V., 2008. Origin of groundwater salinity and hydrogeochemical processes in a confined coastal aquifer: Case of the Rhône delta (Southern France). Appl. Geochemistry 23, 2337–2349. https://doi.org/10.1016/j.apgeochem.2008.03.011

Essink, L., 2018. The interrelationship of salinity and land use in the Khulna Division, Bangladesh, Master Thesis. Utrecht University

Fernández, S., Santín, C., Marquínez, J., Álvarez, M.A., 2010. Saltmarsh soil evolution after land reclamation in Atlantic estuaries (Bay of Biscay, North coast of Spain). Geomorphology 114, 497–507. https://doi.org/10.1016/j.geomorph.2009.08.014

Geelen, L.H.W.T., Kamps, P.T.W.J., Olsthoorn, T.N., 2017. From overexploitation to sustainable use, an overview of 160 years of water extraction in the Amsterdam dunes, the Netherlands. J. Coast. Conserv. 21, 657–668. https://doi.org/10.1007/s11852-016-0452-x

Gleeson, T., Wada, Y., Bierkens, M.F.P., Van Beek, L.P.H., 2012. Water balance of global aquifers revealed by groundwater footprint. Nature 488, 197–200. https://doi.org/10.1038/nature11295

Goes, B.J.M., Oude Essink, G.H.P., Vernes, R.W., Sergi, F., 2009. Estimating the depth of fresh and brackish groundwater in a predominantly saline region using geophysical and hydrological methods, zeeland, the netherlands. Near Surf. Geophys. 7, 401–412. https://doi.org/10.3997/1873-0604.2009048

Groen, J., Velstra, J., Meesters, A.G.C.., 2000. Salinization processes in paleowaters in coastal sediments of Suriname: evidence from δ37Cl analysis and diffusion modelling. J. Hydrol. 234, 1–20. https://doi.org/10.1016/S0022-1694(00)00235-3

Hasan, M. B., Driessen, P. P. J., Majumder, S., Zoomers, A., & Laerhoven, F. Van. (2019). water Factors A ff ecting Consumption of Water from a Newly Introduced Safe Drinking Water System : The Case of Managed Aquifer Recharge (MAR) Systems in Bangladesh. Water, (11), 14.

Hengl, T., De Jesus, J.M., Heuvelink, G.B.M., Gonzalez, M.R., Kilibarda, M., Blagotić, A., Shangguan, W., Wright, M.N., Geng, X., Bauer-Marschallinger, B., Guevara, M.A., Vargas, R., MacMillan, R.A., Batjes, N.H., Leenaars, J.G.B., Ribeiro, E., Wheeler, I., Mantel, S., Kempen, B., 2017. SoilGrids250m: Global gridded soil information based on machine learning, PLoS ONE. https://doi.org/10.1371/journal.pone.0169748

Hoque, M.A., Hoque, M.M., Ahmed, K.M., 2007. Declining groundwater level and aquifer dewatering in Dhaka metropolitan area, Bangladesh: Causes and quantification. Hydrogeol. J. 15, 1523–1534. https://doi.org/10.1007/s10040-007-0226-5

Jeuland, M., Harshadeep, N., Escurra, J., Blackmore, D., Sadoff, C., 2013. Implications of climate change for water resources development in the Ganges basin. Water Policy 15, 26–50. https://doi.org/10.2166/wp.2013.107

Karim, M.R., Manshoven, S., Islam, M.R., Gascon, J.A., Ibarra, M., Diels, L., Rahman, M.M., 2013. Assessment of an urban contaminated site from tannery industries in Dhaka city, Bangladesh. J. Hazardous, Toxic, Radioact. Waste 17, 52–61. https://doi.org/10.1061/(ASCE)HZ.2153-5515.0000139

Leung, C.M., Jiao, J.J., 2006. Heavy metal and trace element distributions in groundwater in natural slopes and highly urbanized spaces in Mid-Levels area, Hong Kong. Water Res. 40, 753–767. https://doi.org/10.1016/j.watres.2005.12.016

McArthur, J.M., Sikdar, P.K., Hoque, M.A., Ghosal, U., 2012. Waste-water impacts on groundwater: Cl/Br ratios and implications for arsenic pollution of groundwater in the Bengal Basin and Red River Basin, Vietnam. Sci. Total Environ. 437, 390–402. https://doi.org/10.1016/j.scitotenv.2012.07.068

McGranahan, G., Balk, D., Anderson, B., 2007. The rising tide: assessing the risks of climate change and human settlements in low elevation coastal zones. Environ. Urban. 19, 17–37. https://doi.org/10.1177/0956247807076960

Mirza, M.M.Q., 1998. Diversion of the Ganges water at Farakka and its effects on salinity in Bangladesh. Environ. Manage. 22, 711–722. https://doi.org/10.1007/s002679900141

Mosler, H. J., Blöchliger, O. R., & Inauen, J. (2010). Personal, social, and situational factors influencing the consumption of drinking water from arsenic-safe deep tubewells in Bangladesh. Journal of Environmental Management, 91(6), 1316–1323. https://doi.org/10.1016/j.jenvman.2010.02.012

Mukherjee, A., Fryar, A.E., Thomas, W.A., 2009. Geologic, geomorphic and hydrologic framework and evolution of the Bengal basin, India and Bangladesh. J. Asian Earth Sci. 34, 227–244. https://doi.org/10.1016/j.jseaes.2008.05.011

Mulligan, M., Cruz, L.L.S., Pena-Arancibia, J., Pandey, B., Mahé, G., Fisher, M., 2011. Water availability and use across the Challenge Program on Water and Food (CPWF) basins. Water Int. 36, 17–41. https://doi.org/10.1080/02508060.2011.543801

Nicholls, R.J., Cazenave, A., 2010. Sea-Level Rise and Its Impact on Coastal Zones. Science (80-.). 328, 1517–1520. https://doi.org/10.1126/science.1185782

Nogueira, G., Stigter, T.Y., Zhou, Y., Mussa, F., Juizo, D., 2019. Understanding groundwater salinization mechanisms to secure freshwater resources in the water-scarce city of Maputo, Mozambique. Sci. Total Environ. 661, 723–736. https://doi.org/10.1016/j.scitotenv.2018.12.343

Oude Essink, G.H.P., Van Baaren, E.S., De Louw, P.G.B., 2010. Effects of climate change on coastal groundwater systems: A modeling study in the Netherlands. Water Resour. Res. 46, 1–16. https://doi.org/10.1029/2009WR008719

Pitt, R., Clark, S., Field, R., 1999. Groundwater contamination potential from stormwater infiltration practices. Urban Water 1, 217–236.

Post, V.E.A., Van der Plicht, H., Meijer, H.A.J., 2003. The origin of brackish and saline groundwater in the coastal area of the Netherlands. Geol. en Mijnbouw/Netherlands J. Geosci. 82, 133–147. https://doi.org/10.1017/S0016774600020692

Rahman, M.A., Wiegand, B. a., Badruzzaman, a. B.M., Ptak, T., 2013. Hydrogeological analysis of the upper Dupi Tila Aquifer, towards the implementation of a managed aquifer-recharge project in Dhaka City, Bangladesh. Hydrogeol. J. 21, 1071–1089. https://doi.org/10.1007/s10040-013-0978-z

Santos, I.R., Eyre, B.D., Huettel, M., 2012. The driving forces of porewater and groundwater flow in permeable coastal sediments: A review. Estuar. Coast. Shelf Sci. 98, 1–15. https://doi.org/10.1016/j.ecss.2011.10.024

Schmorak, S., Mercado, A., 1969. Upconing Fresh Water-Sea Water Interface Below Pumping. Water Resour. Res. 5.

Shahid, S., Hazarika, M.K., 2010. Groundwater drought in the northwestern districts of Bangladesh. Water Resour. Manag. 24, 1989–2006. https://doi.org/10.1007/s11269-009-9534-y

Smidt, JD., 2018. Hydrogeological Study of the Assasuni Region in Coastal Southwest Bangladesh – A combined fieldwork and modelling approach to understand changing flow directions, Master Thesis. Utrecht University

Stuyfzand, P.J., 1993. Hydrochemistry and hydrology of the coastal dune area of the Western Netherlands, Ph.D. thesis. Vrije Universiteit of Amsterdam.

Syvitski, J.P.M., Saito, Y., 2007. Morphodynamics of deltas under the influence of humans. Glob. Planet. Change 57, 261–282. https://doi.org/10.1016/j.gloplacha.2006.12.001

Tran, L.T., Larsen, F., Pham, N.Q., Christiansen, A. V., Tran, N., Vu, H. V., Tran, L. V., Hoang, H. V., Hinsby, K., 2012. Origin and extent of fresh groundwater, salty paleowaters and recent saltwater intrusions in Red River flood plain aquifers, Vietnam. Hydrogeol. J. 20, 1295–1313. https://doi.org/10.1007/s10040-012-0874-y

Van Breukelen, B.M., Appelo, C.A.J., Olsthoorn, T.N., 1998. Hydrogeochemical transport modeling of 24 years of Rhine water infiltration in the dunes of the Amsterdam Water Supply. J. Hydrol. 209, 281–296. https://doi.org/10.1016/S0022-1694(98)00105-X

Van Der Meij, J.L., Minnema, B., 1999. Modelling of the effect of a sea-level rise and land subsidence on the evolution of the groundwater density in the subsoil of the northern part of the Netherlands. J. Hydrol. 226, 152–166. https://doi.org/10.1016/S0022-1694(99)00150-X

Van Halem, D., Bakker, S.A., Amy, G.L., Van Dijk, J.C., 2009. Arsenic in drinking water: A worldwide water quality concern for water supply companies. Drink. Water Eng. Sci. 2, 29–34. https://doi.org/10.5194/dwes-2-29-2009

van Halem, D., Olivero, S., de Vet, W. W. J. M., Verberk, J. Q. J. C., Amy, G. L., & van Dijk, J. C. (2010). Subsurface iron and arsenic removal for shallow tube well drinking water supply in rural Bangladesh. Water Research, 44(19), 5761–5769. https://doi.org/10.1016/j.watres.2010.05.049

Van Weert, F., Van der Gun, J., Reckman, J.W.T.M., 2009. Global overview of saline groundwater occurrence and genesis. IGRAC Rep. GP 2009-1 105.

Wagner F., Tran V.B., Renaud F.G. (2012) Groundwater Resources in the Mekong Delta: Availability, Utilization and Risks. In: Renaud F., Kuenzer C. (eds) The Mekong Delta System. Springer Environmental Science and Engineering. Springer, Dordrecht

Walraevens, K., Cardenal-Escarcena, J., Van Camp, M., 2007. Reaction transport modelling of a freshening aquifer (Tertiary Ledo-Paniselian Aquifer, Flanders-Belgium). Appl. Geochemistry 22, 289–305. https://doi.org/10.1016/j.apgeochem.2006.09.006

Wang, Y., Jiao, J.J., 2012. Origin of groundwater salinity and hydrogeochemical processes in the confined Quaternary aquifer of the Pearl River Delta, China. J. Hydrol. 438–439, 112–124. https://doi.org/10.1016/j.jhydrol.2012.03.008

Ward, J.D., Simmons, C.T., Dillon, P.J., Pavelic, P., 2009. Integrated assessment of lateral flow, density effects and dispersion in aquifer storage and recovery. J. Hydrol. 370, 83–99. https://doi.org/10.1016/j.jhydrol.2009.02.055

Werner, A.D., Simmons, C.T., 2009. Impact of sea-level rise on sea water intrusion in coastal aquifers. Ground Water 47, 197–204. https://doi.org/10.1111/j.1745-6584.2008.00535.x

Williams, D.D., Williams, N.E., Cao, Y., 2000. Road salt contamination of groundwater in a major metropolitan area and development of a biological index to monitor its impact. Water Res. 34, 127–138. https://doi.org/10.1016/S0043-1354(99)00129-3

SUMMARY

In coastal areas, groundwater resources are under pressure. To ensure the sustainable provision of safe drinking water under these increasing pressures, groundwater resource management is essential. An important technology to increase the sustainably available groundwater is Managed Aquifer Recharge (MAR). In southwestern Bangladesh, drinking water provision is problematic, leading to the consumption of bacterial-polluted pond water or the consumption of arsenic or brackish groundwater. To overcome these drinking water problems, MAR has been piloted in the form of 99 small-scale, community-run systems.

This thesis describes the research that focusses on the potential of MAR systems being successful in the hydrogeologically highly-variable data-poor region of southwestern Bangladesh. This potential is approached from a technical and a social point-of-view. A large part of the study focusses on elucidating the previously largely unknown hydrogeological situation in the region, in specific related to the variation in groundwater salinity.

The research is presented according to the following topics in chapter 2-5:

- Local scale study on the hydrogeological and groundwater salinity variation
- Regional study on the variation in shallow (< 60 m) groundwater salinity
- Interview study on the attachment of the local people to their current water options.
- Mapping study on regional differences in potential for MAR

Groundwater salinity variation at a local scale. Chapter 2 presents the results of the research that aimed to determine the palaeo and present-day hydrological processes and the geoscientific controls that determine the large variation in groundwater salinity at a local scale. The study was based on fieldwork to collect high-density hydrological and lithological data along a 5 km transect. The results of the study show the large variation in salinity observed may be explained by a variety of palaeo and present-day hydrological processes. The groundwater in aquifers under thick surface clay layers is controlled by the palaeohydrological conditions prevailing when the aquifers were buried. Groundwater in aquifers under thin surface clay layers is affected by present-day processes, which vary depending on present-day surface elevation. Recharge by rain and rainfed ponds is present in slightly

higher-lying areas and, therefore, fresh groundwater is located at shallow depths. In contrast, the lower-lying areas with a thin surface clay layer have brackish–saline groundwater at shallow depth because of flooding by marine-influenced water, subsequent infiltration and aquifer intrusion.

Groundwater salinity variation on a regional scale. Chapter 3 presents the results of a regional study on the variation in shallow (< 60 m) groundwater salinity. The aim of the study was to assess the regional salinity variation as a function of landscape features and associated hydrological processes. The spatial variation in groundwater salinity was assessed using 442 EC measurements from previous studies and 1998 newly collected EC measurements, which were correlated with well location data (latitude, longitude and depth of the filter) and landscape characteristic data (elevation, soil type, land use and surface clay thickness). Additionally, a geomorphological analysis of landscape features was performed to infer associated hydrological processes. Wide fluvial zones were interpreted to be remnants of sandy deposits in large paleo channels which allow or allowed freshwater recharge, resulting in groundwater that is mostly (75%) fresh. Narrow fluvial zones, tidal fluvial zones, and fluvial zones next to tidal rivers are more susceptible to lateral inflow of saline water or saline water recharge by occasional tidal flooding, and only contain some shallow fresh groundwater in high-lying zones as formed by freshwater recharge. Tidal flat or tidal fringe zones are influenced by saline water recharge, and only contain some fresh groundwater formed under palaeo conditions.

Attachment to unsafe water options. Chapter 4 presents the results of an interview study on the attachment of the local people to their current water options. The aim of the study was to determine to what extent and why people are expected to remain attached to unsafe water options, pond and shallow tube wells (STWs), compared to four safer drinking water options, deep tubewells (DTWs), pond sand filters (PSFs), vendor water and rainwater harvesting (RWH). The results show that users' attachment to pond water is generally low. Factors facilitating a switch towards safer alternatives are perceived risk and attitude, whereas norms are a barrier for such a switch. For STW, reliability and habit are barriers preventing a switch to safer options, and there are no clear facilitators for such a switch. STW users are more attached to their source than pond users, and, therefore, safer drinking water options have less potential to replace STWs than to replace pond water. The safe alternatives DTWs, RWH, PSFs and vendor water score significantly better than pond water, and are estimated to have some potential to be adopted by pond water users; RWH, DTWs and PSFs also score better than STWs, and could have potential to replace STWs.

Regional MAR potential based on necessity and technical suitability. Chapter 5 presents the research on the potential for MAR throughout southwestern Bangladesh. The aim of the study used an approach that combines 1.) the necessity of local communities for MAR based on the groundwater quality, and 2.) the technical MAR suitability based on constraints for MAR and the expected effect of density-driven flow and vulnerability to mixing. The results show that there generally is a mismatch between the necessity and technical suitability. In parts of the north of the region, suitable groundwater is present, causing there to currently be no necessity for MAR to be implemented, despite the high technical suitability. In other parts of the north of the region, arsenic or brackish groundwater is likely to be consumed, while MAR is generally expected to perform well technically, although there could be a lack of acceptance. In southern parts of the region, people being most likely to have to resort to consumption of pond water, causing to be a primary necessity and potential for MAR to be adopted. However, the low technical suitability calls for careful evaluation of the MAR design or capacity.

MAR implementation in southwestern Bangladesh. Based on the findings in chapter 2 to 5, several implications for the MAR implementation in southwestern Bangladesh are discussed. First, the system-based conceptual understanding of the hydrogeological situation, groundwater salinity, and steering palaeo and present-day processes provided in chapter 2 and 3 can be used during local site-selection to provide a detailed estimate of the groundwater salinity and to assess possible present-day hydrological processes that could influence the MAR systems during operation. Second, the results of chapter 4 can be used to estimate and improve the likeliness of MAR systems being accepted as source for drinking water in favour of ponds and STWs. This can be enabled by ensuring the reliability of the MAR systems, evaluating the distribution strategy to improve the attitude of the community towards MAR, by informing and involving the community during MAR implementation to shape the risk perception and norms, and to ensure the price of MAR is not a hurdle compared to pond or STW water. Third, the results of chapter 5 can be used to identify opportunities for MAR, and whether implementation should focus on improving the acceptance of MAR or on overcoming anticipated technical challenges. However, there are also other important factors during MAR implementation, such as the water source for MAR, possible clogging of the MAR systems, and the scale of the MAR and size of the community.

Relevance for coastal regions worldwide. The findings and methodology of the research are to some extent relevant for other coastal regions where MAR could be applied to overcome salinity or other water quality issues. The lessons and applied methodological framework from this thesis are valuable when trying to understand and predict the groundwater salinity variation in coastal regions with a large salinity variation but limited available data. The results of the research on attachment are only limitedly applicable in these other regions, as there may be different water options available, and the perception of those water options can vary largely between regions, due to differences in cultural and socioeconomical situations. The results of the MAR potential mapping reveal that also in other coastal regions, a possible mismatch between necessity and technical suitability for MAR can be anticipated. However, the results and conclusions of such mapping exercises largely depend on the local conditions in other coastal areas, such as the available water resources, the amount of lateral flow and the local lithological situation.

NEDERLANDSE SAMENVATTING

Drinkwatervoorziening staat in kustgebieden wereldwijd onder druk door onder andere bevolkingsgroei, toenemende welvaart, klimaatverandering, en zoutwaterintrusie. Het goed beheren van de huidige watervoorraden in deze gebieden is essentieel om duurzame voorziening van veilig drinkwater te garanderen. In toenemende mate is ondergrondse zoetwaterberging (Managed Aquifer Recharge; MAR) hierbij een belangrijke technologie. In het kustgebied van zuidwest Bangladesh is de voorziening van schoon en veilig drinkwater problematisch, waardoor er vaak bacterieel verontreinigd oppervlaktewater of grondwater met een schadelijk hoog arseen- of zoutgehalte wordt gedronken. Om dit te verhelpen was in dit gebied ondergrondse zoetwaterberging geïntroduceerd middels het plaatsen van 99 testsystemen die door de lokale gemeenschappen worden beheerd. Complicaties zijn hierbij opgetreden die zowel technisch als sociaal van aard waren. Om deze complicaties te onderzoeken is een wetenschappelijk onderzoeksproject ‘DeltaMAR’ gestart, waarvan dit proefschrift onderdeel is. Dit project werd uitgevoerd door een consortium van Utrecht Universiteit, Delft University of Technology, Dhaka University en Acacia Water en werd gefinancierd door NWO-WOTRO.

Dit proefschrift beschrijft sociaal-hydrogeologisch onderzoek naar de potentie van ondergrondse zoetwaterberging in zuidwest Bangladesh: een regio met een grote hydrogeologische variatie waarover weinig geo-informatie beschikbaar was (en welbeschouwd nog steeds is). Deze potentie is van een technisch-hydrogeologische en een sociale kant benaderd. Zowel de technische als de sociale kant wordt gestuurd door de hydrogeologie. Daarom richt een groot deel van het proefschrift op het verhelderen van de voorheen onbekende hydrogeologische variatie in het gebied, met een specifieke focus op de variatie in zoutgehalte van het grondwater.

In hoofdstuk 2 tot en met 5 zijn de volgende afzonderlijke componenten van het onderzoek beschreven:

- Hoofdstuk 2: Variatie in hydrogeologie en zoutgehalte van het grondwater op lokale schaal voor een gebied rondom een dorp in Upazilla Assasuni
- Hoofdstuk 3: Variatie in het zoutgehalte van het ondiep (<60 m) grondwater op regionale schaal voor de districten Sathkira, Khulna en Bagerhat
- Hoofdstuk 4: Interviews over de mate waarin mensen vasthoudend zijn in

het gebruik van hun huidige, meestal niet-veilige drinkwatervoorziening

- Hoofdstuk 5: Ontwikkeling en toepassing van een methodiek om regionale verschillen in het potentieel voor ondergrondse zoetwaterberging te herleiden op basis van sociale behoefte en technische haalbaarheid

Lokale variatie in het zoutgehalte van het grondwater. In hoofdstuk 2 worden de resultaten gepresenteerd van het onderzoek naar de paleohydrologische en actuele hydrologische processen en de sturende geowetenschappelijke factoren die het zoutgehalte van het grondwater op een lokale schaal bepalen. Voor deze studie is hydrologische en lithologische data in hoge dichtheid verzameld langs een 5 km langprofiel in Upazila Assasuni. De resultaten van de studie tonen een grote variatie in zoutgehalte die kan worden verklaard door verschillende paleohydrologische en actuele hydrologische processen. Onder dikke deklagen van klei is de grondwatersituatie bepaald door paleohydrologische omstandigheden op het moment dat de aquifer werd begraven. Onder dunne deklagen van klei daarentegen wordt het grondwater juist beïnvloed door huidige hydrologische processen, die variëren met de hoogte van het maaiveld ook al zijn deze verschillen klein. Infiltratie van regen en zoet oppervlaktewater uit aangelegde vijvers vindt plaats in de hoger gelegen delen, waardoor er zoet grondwater voorkomt. In de lager gelegen delen met een dunne kleideklaag is brak of zout grondwater gevormd door overstromingen met zout water, gevolgd door infiltratie.

Regionale variatie in het zoutgehalte van het grondwater. In hoofdstuk 3 worden de resultaten gepresenteerd van de regionale studie over de variatie van het ondiep (< 60 m) grondwaterzoutgehalte als een functie van landschappelijke kenmerken en bijbehorende hydrologische processen. De ruimtelijke variatie van het grondwaterzoutgehalte is vastgesteld door middel van 442 metingen van het elektrische geleiding van het grondwater (Electrical Conductivity; EC) van voorgaande studies en 1998 nieuw verzamelde EC metingen. Deze metingen werden gecorreleerd met locatie data (lengtegraad, breedtegraad en diepte) en landschappelijke kenmerken (maaiveldhoogte, bodemtype, landgebruik en dikte van de deklaag van klei). Daarnaast is een geomorfologische analyse van landschappelijke kenmerken uitgevoerd om bijbehorende hydrologische processen af te leiden. Brede fluviatiele zones liggen relatief hoog in het landschap en zijn geïnterpreteerd als overblijfselen van zandafzettingen door grote paleorivieren, waar infiltratie van zoetwater plaatsvindt of -vond. Hierdoor is het grondwater binnen deze zones voornamelijk zoet (75%). Smalle fluviatiele zones, getijde-fluviatiele zones, en fluviatiele zones naast getijderivieren, liggen ook relatief hoog in het landschap, maar zijn gevoeliger voor laterale zoutwaterindringing van nabije getijderivieren of infiltratie van zoutwater tijdens getijdeoverstromingen. Hierdoor bevindt zich binnen deze drie zones minder zoet grondwater. Getijde vlaktes of zones op de getijdenrand liggen lager in het landschap, en zijn beïnvloed door infiltratie van zoutwater, waardoor zoet grondwater weinig voorkomt en dan alleen als het onder paleo-omstandigheden was geïnfiltreerd.

Vasthoudendheid in gebruik van onveilige drinkwatervoorzieningen. In hoofdstuk 4 worden de resultaten gepresenteerd van de interviews naar de vasthoudendheid van de lokale bevolking in zuidwest Bangladesh aan hun huidige vorm van drinkwatervoorziening. Het doel van de studie was om te bepalen in welke mate en waarom de lokale bevolking vasthouden aan onveilige drinkwatervoorzieningen (vervuild oppervlaktewater uit vijvers en ondiepe putten verontreinigd met zout of arseen) vergeleken met vier veiligere opties (diepe putten, oppervlaktewaterzandfilters, water van verkopers, en regenwater). De resultaten laten zien dat de gehechtheid aan oppervlaktewater over het algemeen laag is, waarbij de perceptie van de risico's en de opvatting van de mensen over het oppervlaktewater een overstap naar een veiligere optie kunnen faciliteren, terwijl de sociale normen van de lokale gemeenschap een overstap kunnen belemmeren. Voor ondiepe putten zijn er geen faciliterende factoren, terwijl betrouwbaarheid dat de put water verschaft en gewoontes van de gebruikers belemmerend kunnen zijn voor een overstap. Gebruikers van ondiepe grondwaterputten zijn dus meer gehecht aan hun drinkwateroptie dan gebruikers van oppervlaktewater, maar zulke putten kunnen wel grondwater oppompen met hoge arseen concentraties. Hierdoor hebben veiligere opties minder potentie om ondiepe putten als drinkwatervoorziening te vervangen dan om oppervlaktewater als bron te vervangen. Alle vier bestudeerde veilige alternatieven scoren beter in de interviews dan oppervlaktewater als bron en hebben naar verwachting potentie om het gebruik van oppervlaktewater als bron van drinkwater te vervangen. Drie van de vier veilige alternatieven, namelijk regenwater, diepe putten en oppervlaktewaterzandfilters, hebben ook een betere score dan ondiepe putten, waardoor ze naar verwachting ook potentie hebben om ondiepe putten te vervangen als vorm van drinkwatervoorziening.

Regionaal potentieel voor ondergrondse zoetwaterberging, gebaseerd op sociale noodzaak en technische geschiktheid. Hoofdstuk 5 combineert een analyse van 1) de sociale noodzaak van ondergrondse zoetwaterberging voor lokale gemeenschappen, gebaseerd op de kwaliteit van het grondwater als bron van drinkwater, en 2) de technische geschiktheid van het gebied voor ondergrondse zoetwaterberging gebaseerd op landgebruiksfactoren, het verwachte effect van dichtheidsstroming en de kwetsbaarheid voor menging met oorspronkelijk aanwezig grondwater verontreinigd met zout en/of arseen. Tijdens deze analyse wordt gebruik gemaakt van de meest complete database van arseen en zout in het grondwater om regionale verschillen in potentie van ondergrondse zoetwaterberging in zuidwest Bangladesh in kaart te brengen. De resultaten laten zien dat er over het algemeen een discrepantie is tussen de sociale noodzaak en de technische geschiktheid. In sommige delen van het noorden van het studiegebied is veilig grondwater beschikbaar, waardoor er geen noodzaak is van kunstmatige, ondergrondse berging van zoet water, ondanks dat het gebied er technisch geschikt voor is. In andere delen in het noorden komt de consumptie van arseenhoudend of brak grondwater voor en is er daardoor sociale noodzaak voor een veiliger drinkwateroptie. De verwachting is dat ondergrondse zoetwaterberging technisch goed zal werken, hoewel mensen vanwege hun gehechtheid/vasthoudendheid aan de onveilige bronnen mogelijk minder genegen zijn om op de veilige optie over te gaan. In zuidelijke delen van het gebied is een deel van de mensen aangewezen op het drinken van oppervlaktewater uit vervuilde vijvers, waardoor er een hoge sociale noodzaak is voor ondergrondse zoetwaterberging. De lage technische geschiktheid van het gebied vraagt echter om zorgvuldige evaluatie van het ontwerp en de capaciteit van de ondergrondse zoetwaterberging systemen. Specifiek zijn systemen met een groot injectie en extractie debiet minder gevoelig voor dichtheidsstroming en menging met omliggend grondwater. In de zuidelijke delen van het gebied zouden grote, gemeenschappelijke systemen technisch gezien een betere keuze zijn dan de huidige kleinschalige systemen.

Implementatie van ondergrondse zoetwaterberging in zuidwest Bangladesh. Op basis van de bevindingen beschreven in hoofdstuk 2 tot een met 5, zijn verschillende implicaties te formuleren voor de implementatie van ondergrondse zoetwaterberging in zuidwest Bangladesh. Ten eerste zal het conceptuele systeembegrip van de hydrogeologische situatie (grondwater zoutgehalte, en sturende paleo en huidige hydrologische processen, zoals beschreven in hoofdstuk 2 en 3) kunnen worden gebruikt tijdens de selectie van locaties om ondergrondse zoetwaterberging toe te passen. Specifiek kunnen locaties worden geselecteerd met een lage kans op eventuele invloeden van hydrologische processen, zoals dichtheid stroming of zoutwater infiltratie. Ten tweede zijn de resultaten van hoofdstuk 4 te gebruiken om de kans in te schatten en daarbij ook te vergroten dat de ondergrondse zoetwaterberging systemen door de bevolking daadwerkelijk zullen worden gebruikt. Deze kans kan worden vergroot door water van de systemen rechtstreeks naar de gebruikers te distribueren en de gemeenschap te betrekken tijdens de implementatie. Ten derde kunnen de resultaten van hoofdstuk 5 worden gebruikt om behoefte voor ondergrondse zoetwaterberging te identificeren, en om te bepalen of men bij de implementatie zou moeten focussen op het verbeteren van de kans op acceptatie, op het overwinnen van technische uitdagingen of beide. Naast deze factoren zijn er ook nog andere factoren belangrijk tijdens de implementatie van ondergrondse zoetwaterberging, zoals de beschikbaarheid van water om te bergen, mogelijke verstoppingen van putten in het systeem, en de schaal en grootte van het systeem in relatie tot de doelgemeenschap.

Relevantie voor kustgebieden wereldwijd. De bevindingen en methodes van het onderzoek zoals beschreven in dit proefschrift zijn relevant voor andere kustgebieden met zout- of andere waterkwaliteitsproblemen. De bevindingen en het toegepaste methodologische kader van hoofdstuk 2 en 3 zijn overdraagbaar naar andere kustgebieden met een grote variatie in zoutgehalte, maar met een gelimiteerde beschikbaarheid van data. De bevindingen van het onderzoek naar de vasthoudendheid van mensen in het gebruik van hun drinkwaterbron zijn tot op zekere hoogte toepasbaar in andere gebieden. Wel moet rekening worden gehouden met culturele en socio-economische verschillen en met de beschikbaarheid van andere wateropties.. De resultaten van het in kaart brengen van de potentie voor ondergrondse zoetwaterberging suggereert dat ook in andere kustgebieden een discrepantie kan worden verwacht tussen de sociale noodzaak en de technische geschiktheid van ondergrondse zoetwaterberging, doordat zout grondwater enerzijds de sociale noodzaak verhoogt maar anderzijds de technische prestaties door dichtheidsstroming en menging verminderd. Hierbij moet wel worden genoemd dat deze resultaten en conclusies afhankelijk zijn van lokale omstandigheden, zoals de beschikbare watervoorraden, de hoeveelheid grondwaterstroming, en de lokale hydrogeologische situatie.

DANKWOORD/ACKNOWLEDGEMENTS

Het is inmiddels ongeveer 5 jaar geleden dat ik solliciteerde op de PhD positie in dit project. Ik zat destijds nog in Wenen, waar ik bezig was met mijn master thesis. Dit vond ik erg leuk en ging me vrij gemakkelijk af, dus ik dacht, een PhD lijkt me dan ook erg leuk. Een PhD is echter een stuk lastiger dan een master thesis. Zeker in het begin, met zeer gebrekkige data en met een veldwerkgebied halverwege de andere kant van de wereld, was het lastig om precies te weten hoe ik dit klusje moest gaan klaren. Gelukkig is het sindsdien alleen maar leuker geworden. Zeker op het moment dat ik zelf data mocht verzamelen en verwerken, werd dit echt mijn droombaan. Uiteindelijk is het dan ook gelukt om het voor elkaar te krijgen. Tijdens mijn PhD heb ik veel steun gehad van vele verschillende mensen, zonder wie ik het niet allemaal voor elkaar had gekregen. Hieronder wil ik ze allemaal bedanken.

Als eerst wil ik mijn begeleiders bedanken. Paul Schot was mijn dagelijkse begeleider, waarbij ik altijd binnen kon lopen om te spreken over iets waarbij ik vastzat, of waarbij ik een tweede, scherpe, visie kon gebruiken. De tijd in Bangladesh met Paul kan ik ook niet vergeten: Paul liep erbij als een kind in een snoepwinkel, overal speurend naar waar het vele water in het landschap naartoe stroomde en non-stop bezig met het opperen van ideeën en hypothesissen. Daarnaast heb ik erg veel gehad aan de wandelende bibliotheek Jasper Griffioen. Van Jasper kreeg ik steevast het juiste advies op vragen of onduidelijkheden waarmee ik zat, vaak in de vorm van een stapel artikelen die ik moest lezen. Het was niet zo dat ik altijd zonder vragen wegging na een meeting met Jasper. Nee, juist het omgekeerde: door de uitgebreide kennis van Jasper, kreeg ik een hele hoop nieuwe vragen en dingen die ik nog moet uitzoeken. Uiteindelijk heeft dit ervoor gezorgd dat mijn proefschrift uitgebreid en compleet is geworden, zonder al te veel losse eindjes.

The third supervisor that I had was Kazi Matin Ahmed from Dhaka University. The expert of hydrogeology in Bangladesh gave me much of the initial knowledge and information needed to start forming my research questions. I also owe a large part of the success of the fieldwork campaigns to Matin, as he arranged all the practicalities in the field and always made sure some of his MSc students were interested to join the fieldwork.

Graag wil ik ook de rest van het DeltaMAR team bedanken. Frank van Laerhoven, je leek altijd wel druk te zijn met het schrijven van een of ander rapport voor NWO, of met het voorbereiden van een nieuwe workshop. Ik zal

nooit vergeten hoe we hebben ons hebben uitgesloofd en hebben gezweet tijdens de workshops in Khulna, waarbij ook de external reviewer aanwezig was. Ook heb ik het geluk gehad om samen met je een artikel te kunnen schrijven, waarbij de samenwerking heel erg soepel ging. Boris van Breukelen, ik wil je graag bedanken voor de vele prikkelende inhoudelijke discussies. Doordat je met een nét andere blik kijkt dan Paul en Jasper, kreeg ik steeds nieuwe motivatie. Lastly, I would like to thank Mark Bakker and Shantanu, with whom I have had the pleasure to have several interesting conservations.

Of course, I would also like to thank the other PhD-students of the DeltaMAR project, with whom I could share the struggles of doing a PhD. Risalat, you were a great fieldwork partner in the early stage of my research, and your knowledge of the MAR-project was extremely valuable to me. Badrul, we have seen each other often at the University in Utrecht, which I always enjoyed (although you were making too long hours!). Imran, we have collaborated less, but the moments we were together were always constructive.

Special thanks to the whole MAR-office in Khulna. Thanks to them, I had a place that felt like home in Bangladesh. Abir, you were the driving motor during the fieldwork. Thank you for being a great friend, always ready to help. I will remember our time together in the field forever, and I will also always remember visiting your lovely house in Jessore.

Ook wil ik een aantal mensen van Acacia water bedanken. Acacia water was betrokken bij de voorloper van dit PhD-project, bij het PhD-project zelf, en zetten zich nu nog steeds in voor de MAR-systemen na dit PhD-project. Specifiek wil ik Koos Groen bedanken, van wie ik veel heb geleerd. Daarnaast dank aan Tine, Sieger en Lukas, voor hun visie op en inzet voor de MAR-systemen in Bangladesh.

Het meeste heb ik genoten van de tijd in het veld. Met name in januari-februari 2017 en januari-februari 2018 ik twee brede veldwerk campagnes opgezet. Deze opzet heb ik te danken aan Paul, die me steeds aanspoorde om vooral groots te denken. Voor het veldwerk in 2017, wil ik Rebecca van Weesep, Frank van Broekhoven en Aria Hamann bedanken voor de onvergetelijke tijd. Ik denk nog graag terug aan de tijd daar in het veld en in de Sundarbans, samen met de mooie foto's van Frank. Daar kregen we aanvullende hulp van Atikul, Pavel, Tofik en weer Abir: thank you very much for the invaluable help in the field. Voor het veldwerk in 2018, zijn Jan Dirk Smidt, Kennard Burer, Ruth Wijland en Luc Essink meegegaan. Dit keer kwam ik iets meer in een leidinggevende functie, omdat we met 2 à 3 teams tegelijk in het veld waren. Bedankt voor het veel te hard werken, en voor de goede herinneringen. Dit

keer kregen we hulp van Arif, Mahfuz, Bijoy, Refrat en wederom Abir: thank you for the hard work and excellent company.

Na terugkomst uit het veld, moesten alle monsters (na een soms ingewikkelde reis naar NL), geanalyseerd worden in het lab. Bedankt voor alle hulp van de medewerkers van het milieulab van de UU, in specifiek Thom Claessen en John Visser.

De laatste stap naar publicaties en een dissertatie is het schrijven. In overleg met Paul en Jasper lukte het wel om de inhoud voor elkaar te krijgen, maar academisch schrijven vraagt ook om een uitstekende Engelse tekst. To make that a possibility, I would like to thank Joy Burrough for correcting my many mistakes, and I think all the readers should thank her too!

Ik wil al mijn collega's bedanken van Environmental Sciences. Gilian, bedankt voor het zijn van een geweldige kamergenoot. Doordat we vrij gelijkliepen en dezelfde promotor hadden, kwamen we tegelijkertijd vergelijkbare twijfels en obstakels tegen. Het was dan ook erg fijn om ze te kunnen delen. Feroz and John, thanks for the many pointless discussions during our coffee breaks, and for the very productive times during the shut-up-and-write sessions. Verder nog dank aan alle andere collega's, in ieder geval Iris, Annick, Poornima, Joanke, Rémon, Svenja, Jetske, Mara, Maria, Leontien, Angie, Brian, Dominique, Ann-Hélène, Yasmina, Elizabeth, Myrna, Ineke – sorry als ik iemand vergeet!

Naast het inhoudelijk werk had ik genoeg afleiding van vrienden. Remco, Yolande, Maurits Leenheers en Lilian (inmiddels ook Leenheers), bedankt voor alle gezellige avondjes in Den Bosch, en voor de geweldige tijd skiën, zeilen en kamperen. Bedankt aan alle Soppende Hazen, Menno, Jesse, Simon, Hans, Rex, Stephan, Lynn, Maya, Bernd, Victor en Stijn voor de avondjes in het hooischip, bij Hesp of bij elkaar thuis. Ook bedankt voor de zinloze discussies over onder andere wielrennen en de kades van Amsterdam. Daarnaast nog dank aan Tim Scheffe, Maurits Hortensius en Menno voor de mooie ex-VBU uitjes. Thank you, Manuel and Janitza for keeping in touch and meeting up several times after our unforgettable time in Vienna.

Natuurlijk wil ik ook mijn familie bedanken. Dank je wel mama en papa, voor de steun en motivatie, en voor een huis waar ik altijd welkom ben. Ook bedankt voor jullie pogingen om mijn artikelen te lezen. Pepijn, Ivo en Charlotte, bedankt voor de gezellige avondjes bordspellen, mogen we dat nog lang volhouden. Dan ook nog mijn schoonfamilie bedankt, natuurlijk mahmi (Rosita) en mahma (Ravi) voor het vele lekkere eten en de gastvrijheid, en Ratha die altijd klaar staat voor een goede tijd.

Robia, je betekent echt heel veel voor me. Ik ben zo gelukkig dat ik jouw steun en liefde mag ontvangen. Bij jou kan ik met alles terecht, en dat heb ik ook vaak gedaan tijdens mijn PhD. Inmiddels zijn we al meer dan twee jaar getrouwd, en wordt het nog steeds alleen maar leuker en fijner met je. Ik heb heel veel zin in de toekomst die voor ons in het verschiet ligt. Ik hou ontzettend veel van je.

CURRICULUM VITAE

Floris Naus was born on the 14th of February 1992 in 's-Hertogenbosch, where he grew up and lived until he was 18. He completed his Gymnasium degree in 2010 at the Sint-Janslyceum in 's-Hertogenbosch. In 2011, he moved to Amstelveen to study at the VU University Amsterdam. At the VU, he obtained his Bachelor of Science degree in 2013 after following the bachelor program Earth science and Economics. During this time, Floris also made efforts as the treasurer for the Tenants association of the student campus Uilenstede. After his bachelor, he enrolled in the master's program Hydrology at the VU from which he graduated cum laude in 2015. For his master thesis, he moved to Vienna for six months, where he was affiliated with the TU Wien, and where he researched the behaviour of micropollutants in a riverbank filtration system.

In October 2015, Floris started working on his PhD at the department of Environmental Sciences at the Utrecht University. This PhD was part of the larger DeltaMAR project and focused on assessing the socio-hydrogeological potential for Managed Aquifer Recharge in the fresh-saline aquifers of southwestern Bangladesh. The final outcomes of his PhD research are presented in this dissertation.

After his PhD, Floris decided to focus on gaining experience in practical projects. In January 2020, Floris started working for Witteveen+Bos in Amsterdam.

PUBLICATIONS

Scientific:

- **Naus, F. L.**, Schot, P., Groen, K., Ahmed, K. M., and Griffioen, J., (2019a), Groundwater salinity variation in Upazila Assasuni (southwestern Bangladesh), as steered by surface clay layer thickness, relative elevation and present-day land use, Hydrol. Earth Syst. Sci., 23, 1431–1451, https://doi.org/10.5194/hess-23-1431-2019

- van Driezum, I. H., Derx, J., Oudega, T. J., Zessner, M., **Naus, F. L.**, Saracevic, E., Kirschner, A. K.T., Sommer, R., Farnleitner, A. H., Blaschke, A.P., (2019), Spatiotemporal resolved sampling for the interpretation of micropollutant removal during riverbank filtration, Science of The Total Environment, Volume 649, , Pages 212-223, ISSN 0048-9697, https://doi.org/10.1016/j.scitotenv.2018.08.300.

- **Naus, F.L**., Schot, P., Ahmed, K. M., and Griffioen, J., (2019b), Influence of landscape features on the large shallow groundwater salinity variation in southwestern Bangladesh, J. Hydrol. X 5. https://doi.org/10.1016/j.hydroa.2019.100043

- **Naus, F.L**., Burer, K., van Laerhoven, F., Griffioen, J., Ahmed, K. M., and Schot, P., (2020), Behavioural barriers and facilitators for switching away from unsafe drinking water options in southwestern Bangladesh. Water, 12(2), 342.

- **Naus, F.L**., Schot, P., van Breukelen, B., Ahmed, K. M., and Griffioen, J., (2020) Potential for Managed Aquifer Recharge in southwestern Bangladesh based on social necessity and technical suitability (*Accepted for publication at Hydrogeology journal*)

Other:

- **Naus, F.L.**, (2017), Geohydrologisch onderzoek in een land vol water, Geo. brief jaargang 2017 editie 7, ISSN 1876-231X