Chapter 14

Modelling of natural water treatment systems in India: Learning from the Saph Pani case studies

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14.1 INTRODUCTION

Analytical or numerical models can be used at all stages of natural treatment system (NTS) implementation, from initial planning of individual systems, to upscaling at the watershed scale and to system optimisation so as to reach defined water quantity and quality targets. Such models enable water managers to test diverse scenarios so that they can optimise implementation of NTS's within a watershed (which type? where? how big?). At local scale, models also may be useful for improving any individual NTS by fine-tuning technical options. Generally spoken, they are management tools that help avoid costly, real-size trial-and-error testing of NTS's and also may avoid surprises with respect to the expected impact of NTSs on water quantity and quality of a watershed or aquifer.

Within the Saph Pani project, different types of NTSs (river bank filtration, constructed wetlands and managed aquifer recharge) have been modelled in a large variety of geological and hydro-climatic settings, representative of the Indian subcontinent, thus demonstrating the usability of state-of-the art integrated surface-groundwater flow and transport models as planning and management tools.

Numerical flow or flow and transport modelling has been applied to the following problems within the Saph Pani case studies:

River Bank Filtration (RBF)

- Optimisation of well technology and exploitation schemes assisted by flow modelling for RBF in Haridwar, Uttarakhand
- Contaminant transport/attenuation in an RBF scheme in the urban alluvial aquifer of Yamuna River, New Delhi

Managed Aquifer Recharge (MAR)

- Saline intrusion management and implementation of MAR check dams in the coastal Arani and Koratalaiyar watershed, Chennai, Tamil Nadu
- Behaviour of individual MAR-SAT percolation tanks, Maheshwaram, Telangana

Constructed Wetlands

- Wetland impacts on water balance in the Musi watershed, Hyderabad, Telangana

14.2 MODELLING OF RIVER BANK FILTRATION (RBF)

Direct surface water abstraction from river networks for drinking water and irrigation bears important sanitary risks, both microbial and chemical (pathogens, organic contaminants including emerging substance classes, inorganic major, minor and trace compounds). These risks can be considerably reduced through indirect river water abstraction from wells and boreholes within the accompanying alluvial aquifer. Water pumped from such wells will contain a variable fraction of river water (up to 100%) and recharge coming from direct infiltration of rainwater or from groundwater flow from the piedmont area. The river water fraction, called river bank filtrate, will be naturally purified during the passage through (1) the river bed, often rich in clay minerals and organic matter and (2) the alluvial aquifer. Main processes are (1) physical retention of suspended matter and microbes depending on the porosity of the filtering media, (2) chemical interaction of the migrating water with the aquifer material, notably sorption-desorption processes, ion exchange on clay minerals, dissolution-precipitation reactions and (3) microbiologically mediated processes mainly taking place in contact with biofilms on the aquifer material, in particular biodegradation of organic substances, transformation of organic matter, nitrification-denitrification processes. Those processes all have their proper kinetics and are therefore time-dependent. The purifying action of RBF will largely vary in function depending on the contact time of the migrating water with minerals and biofilms.

In the following sub-chapter we will outline some crucial aspects of RBF, including the determination of key parameters for purification capacity, that is, the mixing proportion of river bank filtrated in the pumped alluvial groundwater and the travel time from the river to the pumping wells (Haridwar case study) as well as the simulation of transport of nitrogen species within the aquifer material, both on lab scale and field scale (Delhi case study).

14.2.1 RBF at River Ganga, Haridwar, Uttarakhand: groundwater flow modelling

Site description

The importance of RBF as a sustainable year-round natural treatment technology for the provision of drinking water to the permanent residents of Haridwar (>225,000) and the highly variable number of pilgrims (at least 50,000 daily, with up to 8.2 million on specific days such as *Kumbh Mela*; Gangwar & Joshi, 2004) for the removal of bacteriological indicators (total coliforms and *E. coli*) and for meeting the dynamic drinking water demand has been highlighted in Chapters 1 and 2.

As of 2013, at least two-thirds, or 59,000 to 67,000 m^3/day (Bartak *et al.*, 2015), of the total raw water for drinking was abstracted from a total of 22 RBF wells with the remainder supplied by deep groundwater abstraction wells. The RBF wells are located on the west-bank of the Ganga River in the North and south part of the city, on Pant Dweep Island and on a narrow stretch of land between the Upper Ganga Canal (UGC) (Figure 14.1). Thus, by virtue of their proximity to the Ganga River and UGC that form natural recharge boundaries, the RBF wells abstract around 40 to 90% bank filtrate (Bartak *et al.*, 2015). The portion of bank filtrate abstracted by the wells located on Pant Dweep Island and further south is greater than those to the north. This is due to their location in an area influenced by the naturally occurring flow of bank filtrate between the UGC and Ganga River due to the difference in hydraulic gradient. The naturally pre-treated RBF water is abstracted from the upper unconfined alluvial aquifer, which is in hydraulic contact with the Ganga River and UGC. The aquifer comprises fluvial deposits of poorly graded sand (0.0075–4.75 mm) beneath which lies a lower layer of silty sand (Dash *et al.*, 2010). After abstraction, the water is disinfected with sodium hypochlorite at the well prior to being distributed to the consumer. Although these wells are relatively shallow (7–10 m deep), they have a large storage capacity due to their large diameter (~10 m). The abstraction from these wells is highly variable (790 to 7,530 m^3/day) and dependent on the season, with higher daily abstractions during monsoon as a result of longer operating hours due to a greater water demand, but also due to an increase in groundwater levels due to greater recharge from the surface water bodies.

The discharge of partially treated sewage and untreated storm water run-off into the Ganga River (and UGC in Haridwar) and its upstream tributaries, as well as large-scale ritualistic bathing, are a source of thermotolerant coliforms (TTC; *E. coli)* present in the surface water. In this context, mean TTC numbers measured in the 22 RBF wells, calculated from long term water quality data (2005–2013), were 18 TTC/100 mL during monsoon and 1 TTC/100 mL during non-monsoon compared with 10^4–10^5 TTC/100 mL in the Ganga, including UGC (Bartak *et al.*, 2015). This highlights the significant removal efficiency of 3.5 to 4.4 $\log_{10}$ of TTC by RBF (Dash *et al.*, 2010).

Problems to be solved

Despite the observed high TTC removal efficiency, TTC counts up to 93 MPN/100 mL were still observed in some RBF wells (Bartak *et al.*, 2015). In this context it has been observed that some RBF wells which are very close to the surface water body show the presence of coliforms, such as wells 27, 2, 16, 40, 42, 17, 21 and 24 (for details see Chapter 2) that are located at a distance of 6–36 m from the UGC and its escape channel.

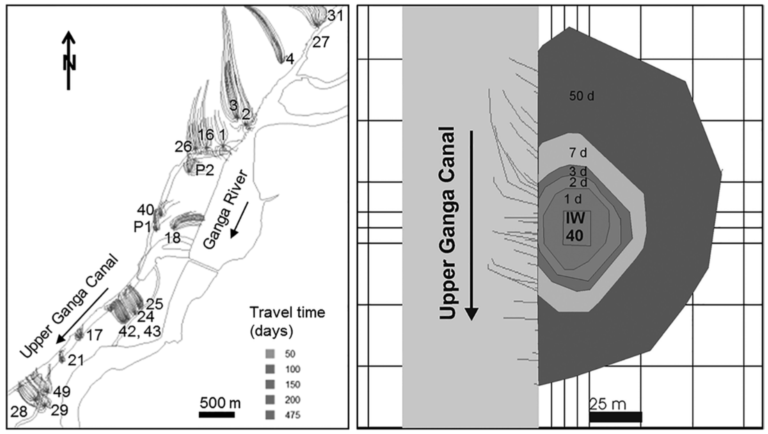

Figure 14.1 Travel time and flow paths of bank filtrate for RBF system in Haridwar (left) and travel time of bank filtrate during monsoon for RBF well 40.

However, it is also evident that RBF wells which are comparatively far away from the Ganga or UGC (48–190 m) and in the area where the Ganga enters Haridwar in the northern part of the city (thus low impact of upstream pollution), such as wells 3, 4, 26, 1 and 31, also have comparably high coliform counts of <2–93 MPN/100 mL (Bartak *et al*., 2015). Normally one would not expect wells to be contaminated at such a relatively far distance. But considering that the lack of well head protection zones, social land use practices such as public bathing/washing at the well heads, well head housing, cattle in and around the RBF wells and unsanitary defecation practices near/around the wells were identified as high risks for the Haridwar RBF system (Bartak *et al*., 2015), it is conceivable that the origin of coliforms in some RBF wells is not the bank filtrate from the Ganga River but rather ambient landside groundwater.

Thus, the objective of the numerical groundwater flow modelling study for Haridwar was to identify the flow paths of the bank filtrate to the RBF wells and the travel time in order to analyse the source of contamination of the wells. Additionally, an overall understanding of the hydrogeological system in response to the dynamic hydrological regime of the Ganga River (high monsoon and low non-monsoon water levels) was to be achieved. The degree of confidence in the numerically simulated portion of bank filtrate abstracted by the RBF wells was to be ascertained through comparisons with analytical calculations from mean electrical conductivity (EC) and Oxygen-18 isotope values.

Tools and modelling strategy

A three-dimensional, finite element, two–layered, numerical groundwater flow model was set up in Visual MODFLOW (version, 2011.1). The spatial extent of the entire model area is 5,000 m (East-West) × 6,000 m (North-South). However, the active model domain is assigned only to the area of the Ganga floodplain with the remaining cells inactivated. The model domain was discretised to obtain a cell area of 12.5 × 12.5 to 100 × 100 m. The upper model layer coincides in general with the upper sandy layer of the aquifer (thickness from 0 m to around 12 m below ground level (bgl) and with the bottom of the partially penetrating RBF wells. The lower layer represents the silty sand (thickness 12 m to around 21 m bgl). The Ganga River, UGC and its escape channel are represented by the river boundary condition. The hydraulic conductivity of the riverbed was determined from sieve analyses at various points and accordingly assigned to calculate the riverbed conductance. Reference day water level measurements were conducted on three specific days, one each in August 2012, October 2012 and January 2013 to represent monsoon, post and pre-monsoon conditions and a relatively good calibration was achieved for each of these conditions. The actual hourly abstraction rates of the RBF wells for the monsoon and non-monsoon operating hours were normalised for a continuous operation (24 hour period) and assigned using the well boundary condition for the 22 RBF wells in the model at their respective locations. Subsequently the particle tracking tool was used in MODPATH to visualize the flow paths and travel times of water to the RBF wells (Figure 14.1). The zone budget method in MODFLOW was used to determine the portion of bank filtrate abstracted by the RBF wells.

Outcome, added value and perspectives

The simulated flow paths of the water to the RBF wells (Figure 14.1, left) corroborate to the portion of bank filtrate abstracted by them that have been calculated from long-term mean EC values (Bartak *et al*., 2015) and Oxygen-18 isotope values (Saph

Pani D1.2, 2013). The flow of water to the wells in Figure 14.1 indicate that the RBF wells located in the northern part of Haridwar also receive a considerable portion of groundwater in addition to some bank filtrate. For wells 3, 4, 26 and 1, the portion of ambient landside groundwater is between 40–60% with the remainder being bank filtrate. Consequently a greater portion of bank filtrate is abstracted in monsoon due to an increase in the Ganga River levels and thereby the water line of the river moves closer to the bank and the wells. But as the area that lies in the groundwater catchment of the RBF wells is densely populated and substantially large, there is a greater risk of contamination from decentralised sewage disposal (septic tanks) and leaky wastewater drains. This would also explain the high TTC counts in the RBF wells in relation to a relatively low portion of bank filtrate.

On Pant Dweep Island the shortest travel time of the bank filtrate to the wells 40 and P1, located only 15 m from the UGC, is around 3 days during the non-monsoon period that decreases to 2 days during monsoon (Figure 14.1, right). The mean portion of bank filtrate abstracted is 60–70% and while the TTC counts in well 40 are <2–93 MPN/100 mL, they are <3 MPN/100 mL or below the detectable limit in well P1 (Bartak *et al*., 2015). Although both wells exhibit short travel times of bank filtrate, bathing and washing activities take place immediately next to well 40 by means of a tap attached to the main distribution pipe at the well. Thus the higher TTC count in well 40 and other wells located close to the UGC bank with similar short travel times, can be explained due to the preferential flow of water into the RBF wells from above ground and around the wells (not river/canal water) due to flooding, intensive rainfalls event or regular seepage / drainage of wastewater from bathing and washing activities (Saph Pani D1.2, 2013). This results in very short travel times (45 minutes to 4.5 hours) as demonstrated by a NaCl tracer experiment on well 40 in Chapter 2 (Sandhu *et al*., 2014). For RBF wells 18 and P2, located between 110 m and 320 m from the UGC and Ganga River, the travel time of the bank filtrate is substantially longer (up to a year, Figure 14.1, left). Compared to well 40, the TTC count is lower with a maximum of only 15 MPN/100 mL. As the bank filtrate to these wells has considerably long travel times, the likely reason is above-ground contamination from wide spread defecation on the vast open spaces of the Pant Dweep Island that has an extremely large influx of pilgrims and tourists daily, especially during festivals like the *Kanwar Mela*. During longer festivals like the *Kumbh* and *Ardh Kumbh Melas*, pilgrims reside on the island for up to 4 months. Unlined pit-latrines are dug for such events that have been assessed as a risk to the wells (Bartak *et al*., 2015).

On the other hand, the remaining wells that are located at a distance of 15 m and more from the UGC in the southern part of Haridwar abstract the highest portion of bank filtrate of all RBF wells in Haridwar (80–90%). The simulated portion of bank filtrate (using the zone budget tool in MODPATH) abstracted by these wells lies within a ±10% confidence limit of the analytically calculated portions using EC and Oxygen-18 isotope data. The maximum TTC count observed in some wells was up to 15 MPN/100 mL while in the others it was below the detectable limit of <3 MPN/100 mL, with the exception of one well having a maximum TTC count of 93 MPN/100 mL (Bartak *et al*., 2015). As the area between these wells and the UGC, its escape channels and Ganga River is not residential, the impact from domestic sewage (septic tanks, pit-latrines) is low. However, occasional high TTC counts can be attributed to washing and bathing activities and inappropriate drainage of water (wastewater, rainfall and/or storm water runoff) near/around the wells.

Most importantly, the comparatively overall low TTC counts highlights the high removal efficiency of the RBF system, because most public bathing takes place daily in this stretch of the UGC from which the bank filtrate to these wells originates. Furthermore, the annual monsoon and the location of the wells in the area result in a natural recharge to the RBF wells thereby ensuring sustainable operation during periods of peak water demand.

Conclusions

The Haridwar RBF system is operating sustainably since 1965. The groundwater flow modelling study of the RBF system in Haridwar has identified the flow paths of bank filtrate and the groundwater catchment areas of the RBF wells. In conjunction with investigations on the risk of floods and health risk assessment to RBF wells using Haridwar as an example (Chapter 2; Sandhu *et al*., 2015; Bartak *et al*., 2015), the groundwater flow modelling investigation has helped to identify potential sources of contamination to the wells. Consequently the study has shown that the wells which abstract the highest portion of bank filtrate, have overall lower or at the most an equal magnitude (only in some cases) of TTC counts compared to RBF wells that abstract an equal or greater portion of ambient groundwater. On one hand the flow modelling study has helped to signify the effectiveness of the natural RBF system to remove pathogens, and on the other hand it illustrates the risk of contamination to unconfined aquifers from inhabited areas without appropriate collection, treatment and discharge of domestic sewage and wastewater. This highlights the need for the implementation of well-head and catchment protection zone measures. These measures have to be prioritised in lieu of the growing pressure on land use and conflicting interests. The flow modelling study has also shown the benefit of locating RBF wells on islands and in areas where a natural flow between surface water bodies occurs to ensure sustainable abstraction. The groundwater flow model of the Haridwar RBF system is a useful tool to compliment the water quality and isotope investigations and can be integrated into a regional hydrogeological assessment of the Haridwar urban area.

14.2.2 RBF at Yamuna River, New Delhi: Ammonium reactive transport modelling

Site description

A further RBF field site investigated as part of Saph Pani is located in New Delhi. The capital city is located in North India in the Indo Gangetic Plain along the banks of the Yamuna River. Within the city the river is dammed by two barrages, one in the north of the city and one in the south. In between both barrages, treated, partially treated, and untreated sewage water feed the river through 16 drains (Government of Delhi, 2006). Numerous production wells draw water from the floodplain aquifer, shallow sand and kankar aquifer made up of river deposits. Due to high groundwater abstraction in the city, losing stream conditions are dominating (Lorenzen *et al.*, 2010) and therefore some of the wells draw a high share of bank filtrate. Through the infiltration of river water, sewage-borne contaminants can enter the aquifer and, depending on their retention and degradation rates in the sediments, can eventually reach the production wells. In this context one parameter of concern is ammonium (drinking water limit: 0.5 mg/L, BIS 10500, 2012).

The Delhi study site comprises a transect of observation points across the flood plain on the East bank of the Yamuna River in central Delhi. It includes several hand pumps and observation wells as well as four Ranney wells (large horizontal collector wells) (Figure 14.2). Because the well field was not specifically designed for RBF, the production wells are not parallel to the river bank but are constructed across the complete width of the undeveloped flood plain. The main focus of the study lies on a Ranney well (P3) located at a distance of 500 m from the river bank. A detailed description of the field site is given in Chapter 4.

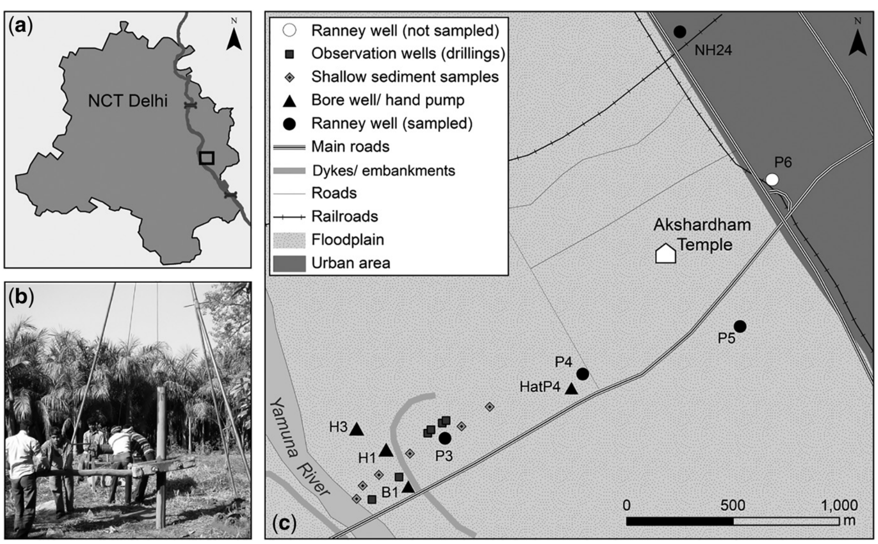

Figure 14.2 (a) Location of the study area (b) Drilling of observation well (c) Location of the hand pumps, Ranney wells and of the shallow and deep drillings conducted during the field work (modified after Groeschke, 2013). Detailed explanation can be found in Chapter 4.

Problems to be solved

The floodplain aquifer is the aquifer with the highest fresh water potential in Delhi (Kumar *et al.*, 2006). However, high ammonium concentrations were found in the river water and in the aquifer close to the river. In the river water ammonium concentrations up to 20 mg/L were measured during the Saph Pani sampling campaigns and concentrations up to 33.3 mg/L were reported by the Central Pollution Control Board (2006). In the sampling points B1, H1, and H3 high variations in ammonium concentrations between 4.5 mg/L and 35 mg/L were found. While 2012 the development of concentrations was similar in the three sampling points, this was not the case in 2013 (Groeschke *et al.*, *submitted a*). In the Ranney well P3, at a distance of 500 m from the river, an increase of ammonium concentrations has been observed for the past years. In 2012 and 2013, the concentrations varied between 5.5 and 8 mg/L and the well was not used for the production of drinking water.

In wells further away from the riverbank, ammonium concentrations remained below 1.7 mg/L in both years. Ammonium concentrations in December 2013 are shown in Figure 14.3. Due to the high ammonium concentrations, the aquifer might be only of limited use for the production of safe drinking water in the future if no appropriate post-treatment or remediation concept is installed. For this, the prediction of future ammonium concentrations is of utmost importance.

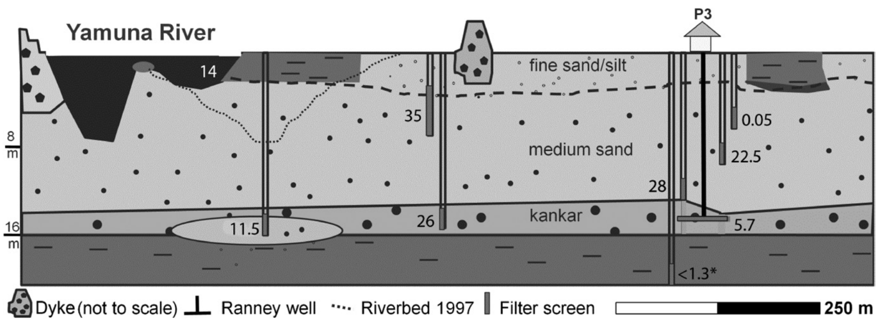

Figure 14.3 Ammonium concentrations in mg/L in December 2013. *Observation well not sampled in December 2013 but all previous concentrations were below 1.3 mg/L. Cross section modified after Groeschke *et al.* (*submitted a*).

Various processes influence the transport and fate of ammonium in an aquifer. Due to interactions with the sediment particle surfaces (cation exchange), ammonium does not move with linear groundwater flow velocity but is retarded. The retardation of ammonium strongly depends on site-specific sediment characteristics and therefore transport cannot be predicted using the retardation factors already published, which vary in magnitudes between 10^0 and 10^2 for sands and gravel, depending on the clay content and the feed concentrations of ammonium (Buss *et al.* 2003). Furthermore, the presence of the reduced nitrogen species ammonium is strongly dependent on the redox conditions in the aquifer. Under oxic conditions it can be biologically oxidised to nitrate in the process of nitrification. Under anoxic conditions it can be also oxidised in the anammox process if nitrate or nitrite are present as electron acceptors (van de Graaf *et al.*, 1995; Clark *et al.*, 2008). In addition, the irreversible fixation of ammonium in clay minerals could also occur as well as the mineralization of organic N as an additional source of ammonium. A detailed description of the redox states, occurrence and effects of nitrogen is given in Chapter 4.

Laboratory column experiments show that fixation or degradation of ammonium takes place to some extent in the sediments of the unsaturated zone and that no degradation takes place in the saturated zone under suboxic and anoxic conditions (see Chapter 4). These results give indications as to the processes occurring at the field site. However, to completely understand the developments of the ammonium concentrations, especially the strong variations, it is necessary to set up a 2D or 3D reactive transport model of the field site. In order to be able to set up such a model, several small scale modelling approaches were applied to determine the necessary input data and to test different hypotheses.

Tools and modelling strategy

The following modelling techniques were applied to gain a better understanding of the processes occurring in the columns and at the field site:

- Inverse geochemical modelling to determine precipitation and dissolution processes occurring during infiltration
- 2D flow and nonreactive transport modelling of column experiments to determine transport parameters of the different lithological units
- 2D and 1D reactive transport modelling of column experiments implementing cation exchange by adapting selectivity coefficients
- 2D and 1D reactive transport modelling of column experiments adding a nonreactive tracer to determine retardation factors
- 1D modelling of two 500 m flow paths

Inverse modelling was conducted with PHREEQC v3 (Parkhurst & Appelo, 2013) to identify reactions which can explain the evolution of the water composition from infiltration to the wells. A sample from the Yamuna River taken in October 2012 was used as the initial water composition and a sample from bore well B1, taken in June 2013, was used as the final solution. Calcite, clay minerals, iron bearing minerals (iron hydroxides, iron sulphides), organic matter, and the exchange species NH_4X, NaX, KX, MgX_2, CaX_2 were included as potential reacting phases (X is a cation exchanger like clay minerals).

A travel time of approximately eight months for the distance of 250 m is in accordance with the average linear groundwater velocity of 0.9 m/d published for this field site (Sprenger, 2011).

A *2D flow and nonreactive transport model* was developed to determine the effective porosities and the dispersivities of the different sediments. The flow simulations were carried out with MODFLOW and the advective-dispersive transport of the NaCl tracer was simulated with the transport simulator MT3DMS and additionally with PHT3D. Tracer breakthrough curves were fitted by adjusting dispersivities and effective porosities, taking into account measured total porosities and literature values (e.g. Johnson, 1967). To ensure that no numerical dispersion or oscillations occurred, the simulations were run with TVD and MMOC solver and selected models were furthermore rerun with smaller grid spacing.

Reactive transport modelling with adapted selectivity coefficients was carried out in 2D and 1D. Using the transport parameters determined with the non-reactive tracer modelling, flow and reactive transport models were developed with MODFLOW and PHT3D to simulate the adsorption and desorption experiments. Many investigations show that at contaminant sites, where the infiltrating water is strongly influenced by one contaminant, simple sorption isotherm models are insufficient to describe the ammonium behaviour at field scale (Buss *et al.*, 2003). Ion exchange models, which consider all species that compete for the exchange sites give better results (Hamann, 2009). Therefore, reactive transport models were developed which consider ion exchange of all the main cations present (Haerens *et al.*, 2002). The reactive transport was computed with PHT3D using the Amm.dat database provided with the software PHREEQC v2. The Amm.dat database decouples ammonium from the nitrogen system, which means that no oxidation of ammonium can occur in the model, which is in accordance with the experimental results showing no oxidation of ammonium to nitrite and nitrite at significant levels (Groeschke *et al.*, *submitted b*). The cation exchange selectivity coefficient is the relative preference of an exchanger to adsorb different cations. It is not a thermodynamic constant, but varies with the exchanger composition (e.g. Tournassat *et al.*, 2007 after Jensen, 1973). For the exchanger phases of the three sediment types (sand, sand with kankar, kankar), equilibrium equations for Na/K, Na/Mg, Na/Ca, Na/NH_4 were set up using the Gaines Thomas convention (Gaines & Thomas, 1953) and measurements of the cation compositions on the exchanger as well as the corresponding activities in groundwater samples.

Retardation factors for ammonium were determined simulating a conservative tracer test by adding a non-reactive, conservative tracer to the reactive transport model (Groeschke *et al.*, *submitted b*). The retardation factors of NH4 were then calculated from the modelled conservative tracer. Also,NH4 breakthrough curves from the time required for the ammonium to reach a relative concentration (C/C_0) of 0.5 at the outlet of the column were compared to the time required for the tracer to reach $C/C_0 = 0.5$ (Steefel *et al.*, 2003), whereby the conservative tracer represents the velocity of the water (Appelo & Postma, 2007 after Sillén, 1951).

1D modelling of flow paths was applied to determine how long it would take to flush out the ammonium from the 500 m wide strip near the river. Detailed description of the model set-up is given in Chapter 4.

Outcome, added value and perspectives

With the help of 2D models, the transport parameters of the two main lithological units of the aquifer (sand and kankar) as well as a transitional unit (sand with kankar) were determined. Reactive transport models were set-up using adapted selectivity coefficients for the different lithological units. This modelling approach gives good results with respect to the development of ammonium concentrations as well as the development of the concentrations of the main cations. In the sand it takes about 10–12 flushes until the 100% ammonium breakthrough is reached and in the kankar it takes about 30–35 flushed pore volumes (Groeschke *et al.*, *submitted b*). The measured and modelled results of one sand and one kankar column experiment are shown in Figure 14.4. Retardation factors were determined by adding a non-reactive tracer to the models; resulting factors are higher than retardation factors published previously for sand and gravel aquifers- between 6.7 and 19.8 (Groeschke *et al.*, *submitted b*) as opposed to between 2.8 and 6.4 (Böhlke *et al.*, 2006). Using the information from all modelling steps, two simplified 1D flow paths from the river were modelled, one in the gravel and one in the kankar layer. Under conditions where only cation exchange occurs and no oxidation of ammonium takes place, it would take 19 years to flush the ammonium from the sand layer and 61 years to flush the ammonium from the kankar layer, provided that the assumed linear flow velocity is accurate.

Conclusions

The enhancement of the river water quality by effective sewage treatment is essential for long term improvement of the groundwater quality in the floodplain aquifer. However, even if this would occur on short term, it would still be a long-term measure due to the strong retardation of ammonium in the aquifer. Therefore, remediation measures or adapted post treatment have to be installed as short and medium term measures, especially as the ammonium concentrations in well P3 will further increase. For this it is essential to know how the ammonium concentrations will develop in the future and if the ammonium plume will reach the drinking water production wells farther away from the river. To obtain more detailed predictions, the information obtained with the modelling techniques described above have to be used to set up a 2D or 3D hydrogeological flow and (reactive) transport models for the floodplain and different source water qualities and pumping scenarios have to be considered.

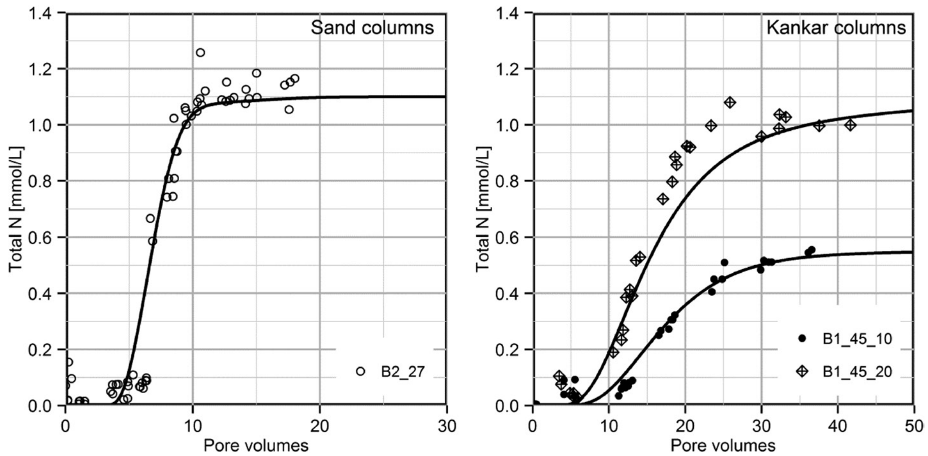

Figure 14.4 Measured (symbols) and modelled (lines) data of selected adsorption experiments. Note the different scales on the x-axes. The breakthrough of ammonium occurs after 12 pore volumes in the sand and after 30–35 pore volumes in the kankar. Modified after Groeschke *et al.* (*submitted b*).

14.3 MODELLING OF MANAGED AQUIFER RECHARGE (MAR)

Managed Aquifer Recharge (MAR) is a method to enhance groundwater quantity and, particularly when combined with Soil-Aquifer-Treatment (SAT), groundwater quality, through the implementation of different types of structures. MAR structures include aquifer storage and recovery (ASR), aquifer storage, transfer and recovery (ASTR), infiltration ponds, infiltration galleries, SAT, percolation tanks and check dams. One important aspect of water quality improvement is also the remediation of saline intrusion into coastal aquifers through increased groundwater recharge upstream in the watershed or through MAR systems (injection well galleries) at the very limit of the salt water wedge.

A thorough understanding of hydrodynamics, at local and watershed scale, is crucial for the selection of the type, dimension and location of MAR structures within a watershed. Flow and transport models, established for those different scales, are important tools to estimate the benefits of MAR-SAT systems for water quality and quantity before implementation and to optimise existing structures. This sub-chapter will look in more details on modelling of a coastal watershed in Tamil Nadu, impacted by over-exploitation and saline intrusion (Chennai case study) and on MAR implementation in a watershed typical for Central India with crystalline fractured bedrock overlain by a more or less porous weathering zone (saprolite).

14.3.1 MAR in a coastal aquifer affected by seawater intrusion: Chennai, Tamil Nadu

Site description

The Arani and Koratalaiyar (A–K) basin is located around 40 km north of Chennai. Surface water from reservoirs and groundwater mostly from well fields of this watershed are one of the major sources for the Chennai city water supply. Excessive and heavy pumping of groundwater from the A–K basin, tidal water ingress, relatively low recharge, generally poor land and water management are the most obvious causes for seawater intrusion. Artificial recharge methods include rainwater harvesting, construction of infiltration wells, percolation tanks, recharge pits and shafts, managing runoff water and facilitating utmost recharge (Asano, 1985). Several check dams were constructed across the Arani and Koratalaiyar rivers to mitigate the problem of seawater intrusion by increasing the groundwater recharge.

The study area comprises two non-perennial river basins Arani-Koratalaiyar (A–K) which are flowing through north of Chennai. The rivers generally flow only for few days during the north east monsoon (November–January). A very dry period occurs in this region during April to May when the temperature rises above 45°C. A colder (winter) period occurs during November to January, experiencing an average temperature of 25°C. The average annual rainfall is around 1200 mm, 35% falling in the south west monsoon (June–September) and 60% during the north east monsoon (October–December). Modelling work has been carried out for an area of 1,455 km^2 in a part of A–K river basin. The Eastern model boundary is delimited by the Bay of Bengal and the south western side is bounded by the Palar River. The elevation in the model area ranges from sea level in the eastern side to 130 m above mean sea level in south west side as observed in the survey of India toposheet. Groundwater has being exploited for the purpose of agricultural and Chennai city water supply. Five well fields were constructed to withdraw groundwater to supply the city with water (Figure 14.5).

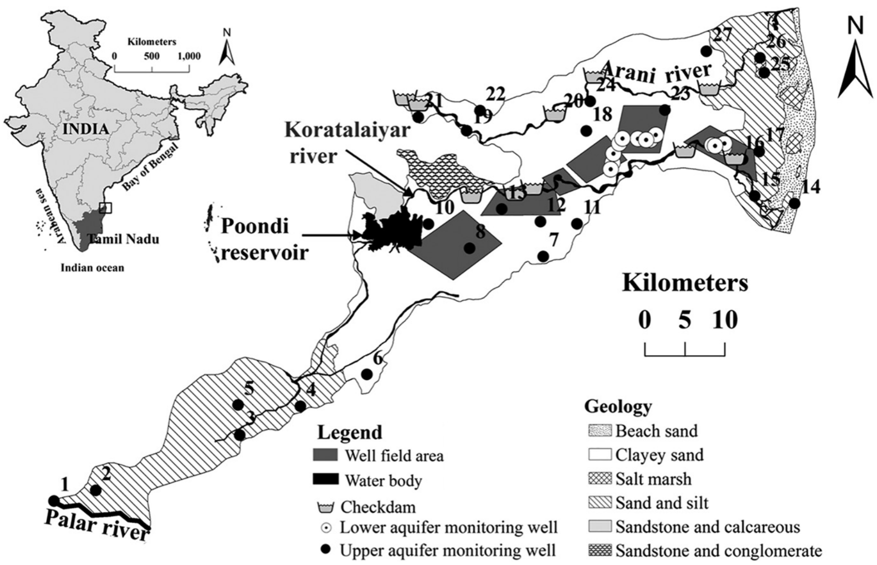

Figure 14.5 Geology of the study area (adapted from Rajaveni *et al.*, 2014a).

Geology and hydrogeology

In this area, the basement Archaean rocks are overlaid by boulders, clay, shale and sandstone of Mesozoic age, the stratigraphic succession of the geologic formations is given in Table 14.1 (UNDP, 1987). The geological outcrop of the A-K basin is shown in Figure 14.5.

Table 14.1 Stratigraphic succession of the geological formation (after UNDP, 1987).

Stratigraphic Age and Thickness	Geological Description
Quaternary (up to 40 m)	Fine to coarse sand, gravel, laterite
Tertiary (45–50 m)	Shale, clay and sand stone
Mesozoic	Gondwana shale and clay
Archaean	Crystalline rocks

The main aquifer in the area is the quaternary alluvium and predominantly consists of fine grained material, reflecting a buried channel system. The subsurface lithology has been characterized by boreholes with depths of 50 m thickness penetrating the coastal alluvium with thicknesses up to 35 m. Groundwater in the area occurs in shallow alluvial zone near the coast and the depth to groundwater level increases with the elevation of the area. The thick clay lenses form a semi confined aquifer system. The groundwater levels in the unconfined aquifer ranges from 2 m to 6 m bgl (below ground level) and in semi confined aquifer it ranges from 14 m to 20 m bgl. A west to east geological cross section is given in Figure 14.6 (A–A′).

Problems to be solved

The A–K basin is characterized by severe over-extraction of groundwater for agricultural activities and water supply to the Chennai metropolitan area, which has been identified to cause significant seawater intrusion (UNDP, 1987). Numerical modelling can help to analyse seawater intrusion by using models to simulate different pumping conditions and quantifying the effect of possible mitigation measures.

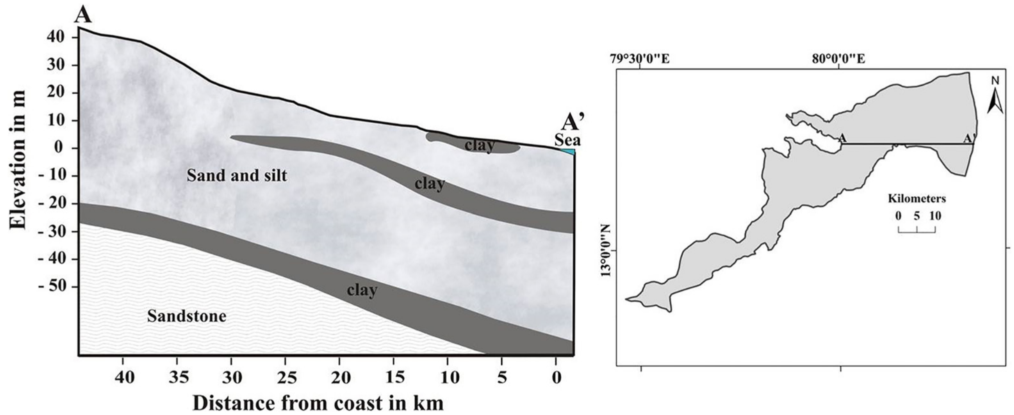

Figure 14.6 Geological cross section along A–A′ line (right figure).

The general objectives are:

- Simulate the current seawater intrusion.
- Representation of check dams and potential other artificial recharge structures in the model to predict future seawater intrusion and to analyse measures to push back the saltwater front.

Tools and modelling strategy

The methods and tools used to generate the coupled model are as follows (Bhola *et al.*, 2014):

1) A rainfall-runoff model (NAM) to produce surface water inflow at the sub-catchment scale as well as the infiltration into the subsoil, integrated in the 1D surface water model.
2) A 1D surface water model (MIKE 11) for the two rivers Arani and Koratalaiyar.
3) A 3D groundwater model (FEFLOW) for the alluvial aquifers of A–K basin which is coupled to the MIKE 11 model using the coupling interface IfmMIKE11 (Monninkhoff, 2011), to describe the interaction between the groundwater and surface water in detail.

Outcome, added value and perspectives

The NAM model parameters were calibrated and extended homogeneously over the entire A–K basin. Since the model was calibrated for an eight year time period, it covers a wide range of hydrologic and climate conditions, which builds confidence in the model's ability to predict stream flow conditions under a variety of scenarios. The model gives a satisfactory comparison with observed flow records with an R^2 value of 0.6. Main focus was given to achieving least volume and peak errors (Figure 14.7). The model over-predicts the total volume in eight years by 10.5%, and a peak error of almost 7% for a discharge greater than 300 m^3/day. The NAM model does not predict low flow accurately due to high surface and root zone storage coefficients. These coefficients define the water holding capacity of the soil, i.e. overland flow will occur once the rainfall is greater than the thresholds of these coefficients. In the observed discharge records, it was found that the response of a rainfall that results in runoff is relatively high and therefore it was implemented accordingly in the model.

Groundwater model The model was calibrated in two stages, steady and transient state condition. The steady state calibration was carried out to achieve an average match between the available observed and simulated groundwater heads and to define a suitable distribution of conductivities. The transient state was carried out for a period of 13 years from January 1996 to March 2009 (Rajaveni *et al.*, *submitted*). Basically transient state calibration was conducted by adapting local conductivities and porosities until the best fit curve was obtained for observed and simulated groundwater heads. A R-square value of 0.901 was obtained during steady state calibration. In the transient state calibration, the simulated groundwater heads were accurately describing the groundwater dynamics of the observed groundwater head in most of the wells. The observed and simulated groundwater head variations in the transient state calibration are exemplarily shown in Figure 14.8 for one of the observation wells.

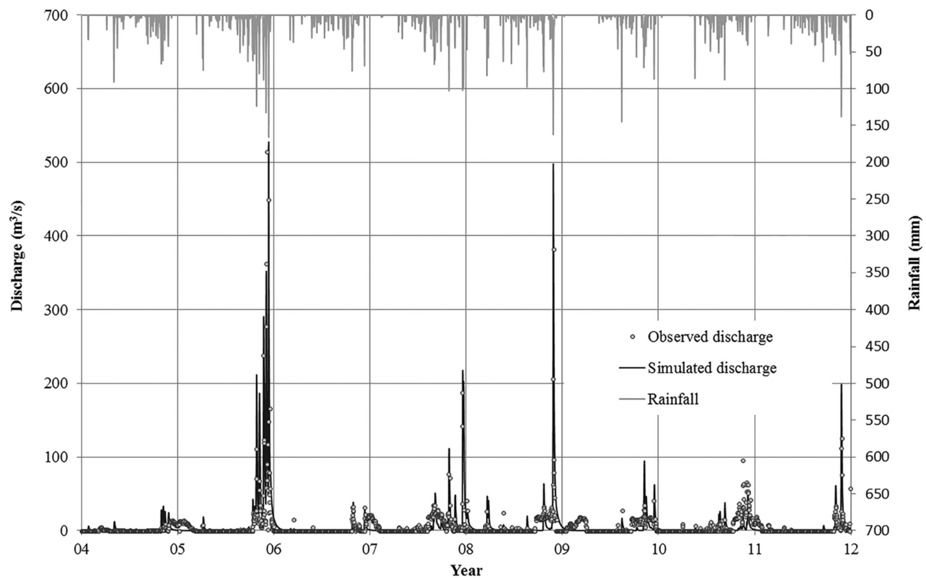

Figure 14.7 Comparison of observed and simulated discharge from 2004 to 2012 at the inlet of Poondi reservoir (Bhola *et al.*, 2014).

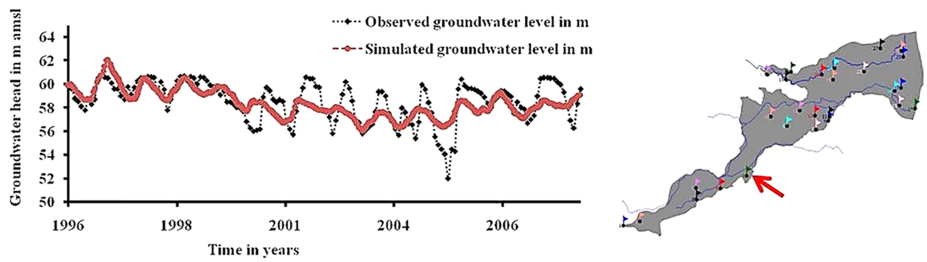

Figure 14.8 Observed and simulated groundwater head variations during transient state calibration in the single aquifer system.

Density dependent model Density dependent parameters were applied to the uncoupled 3D-groundwater model, to report fresh water-seawater interactions. The hydraulic head boundary condition (BC) at the Bay of Bengal was assigned as saltwater head BC. Mass concentration BC was assigned as 500 mg/L in the existing hydraulic head location and 35,000 mg/L in the eastern boundary (coast). An initial mass-concentration distribution was defined, according to the range used in the boundary conditions. To avoid numerical instabilities, the mesh was refined in the coastal region (high density gradient area), which increased the total number of elements from 1 to 1.5 million (Rajaveni *et al.*, *submitted*). An uncoupled density dependent seawater intrusion was simulated and the result shows seawater has intruded from 3.5 km in the year May 1997 to 7 km during May 2003 (Figure 14.9).

Principle simulation of the effect of MAR The general aim of this study is to improve groundwater quantity and quality through MAR structures. As a first step the calibrated 3D groundwater model was used to evaluate the effect of recharge from MAR structures on groundwater heads in the basin. A total of 9 check dams, 4 in the Arani River and 5 in the Koratalaiyar River, existed during 1996 in the study area and were implemented in the uncoupled model. The effect of check dams was computed and predicted by representing the check dams as a fluid transfer BC with different realistic time series (Rajaveni *et al.*, *submitted*). Groundwater head variations were simulated under 2 scenarios (i) with and (ii) without check dams. Observation well 10 has been chosen to explain this study since this observation well is located at the centre of the modelled area. Figure 14.10 shows a maximum of 2 m increase in groundwater heads with the implementation of check dams in the model at this location. The highest differences can be observed during monsoon seasons. During non-monsoon seasons the groundwater head at this location will eventually reach the level representing the situation without check dams.

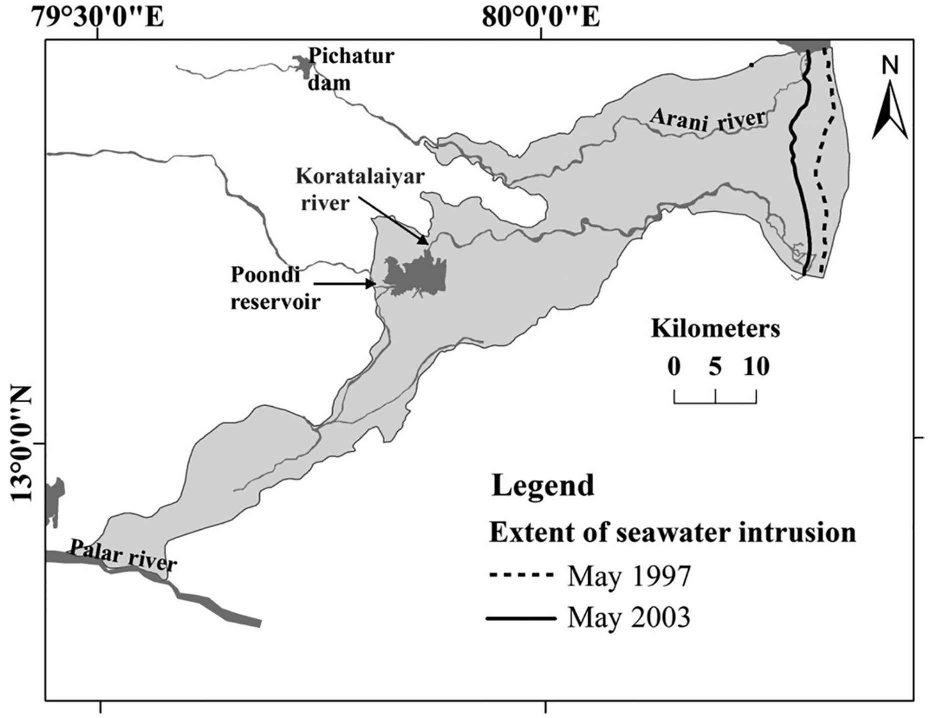

Figure 14.9 Extent of seawater intrusion for two time periods; May 1997 and May 2003 (Rajaveni *et al.*, *submitted*).

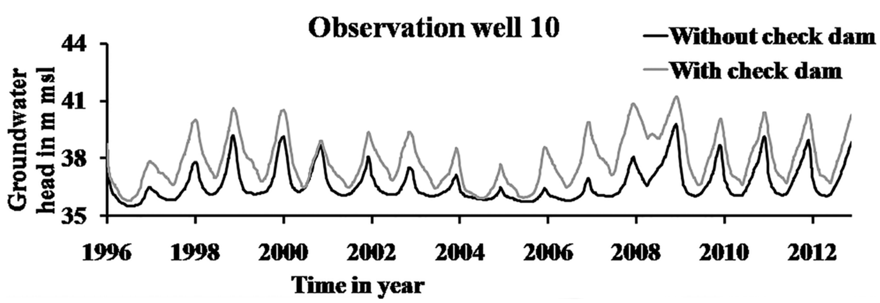

Figure 14.10 Groundwater head variations with and without check dam at the observation well 10 which is located near Poondi.

Coupled surface and groundwater model An uncoupled groundwater flow model was coupled with the surface water model MIKE11 and simulations were performed for the period 2004 to 2009. Three scenarios were simulated: one scenario without check dams, scenario 1 considering most of the existing check dams until present day (9 in total) and scenario 2 with three additional check dams (12 in total) as well as an increased dam crest of about1 m at the already existing check dams (Rajaveni *et al.*, *submitted*).

Figure 14.11 compares the simulated groundwater head along the Arani River, at locations close to the implemented check dams in scenario 2. The last figure (bottom) also includes the simulated water level in the check dam, calculated by the coupled surface water model MIKE11.The following table gives an overview of the situation at 4 selected locations in Figure 14.11. The scenario without check dams represents a situation with approximated natural river courses.

At location A1 there is an existing check dam in scenario 1 and in scenario 2 the same check dam has been raised by 1 m (Rajaveni *et al.*, *submitted*). As expected, the results show an increase in groundwater levels through the implementation of the check dam in scenario 1 and a further, though less significant increase in groundwater heads by raising the dam wall in scenario 2.

Table 14.2 Overview of the scenario definitions at 4 selected locations at the Arani River.

Location	Scenario without Check Dams	Scenario 1	Scenario 2
A1	No check dam	Check dam	Check dam raised
A2	No check dam	No check dam	Check dam implemented
A3	No check dam	Check dam	Check dam raised
A4	No check dam	Check dam	Check dam raised

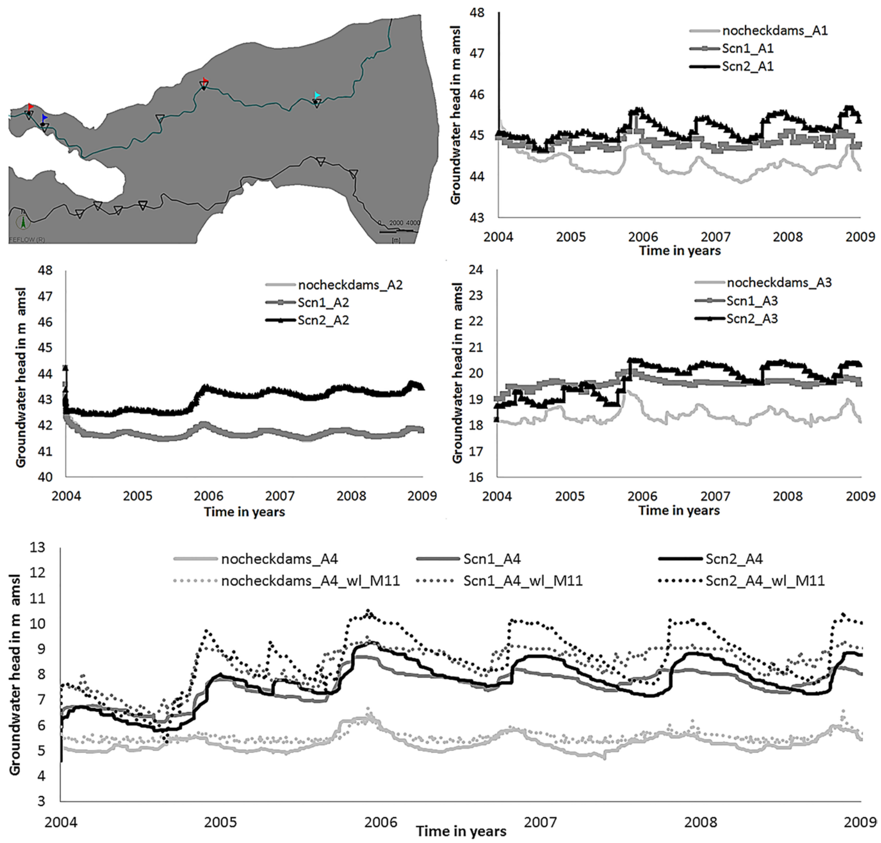

Figure 14.11 Comparison of groundwater heads in the vicinity of existing and plannedcheck damsalongthe Araniyar River for different scenarios (Rajaveni *et al.*, *submitted*).

Scenario 1 has no check dam at location A2, leading to no difference compared to the scenario without check dams. Higher groundwater heads were obtained through the additional check dam in scenario 2.

For location A3 both scenario 1 and 2 have a check dam implemented, while the check dam in scenario 2 has been raised (similar to location A1). The simulation results show that the check dam in scenario 1 increased the groundwater heads and an additional increase in levels can be obtained by constructing a higher check dam wall. At the beginning of the simulation, scenario 1 has slightly higher groundwater heads at this location which is related to the fact that the water is not retained upstream in this scenario since there is no check dam between location A1 and A3 in this simulation. In scenario 2, however, there are two check dams which can retain the water along this river stretch, preventing the check dam at location A3 from being continuously refilled (Rajaveni *et al.*, *submitted*). After 2006, the results show that during wet periods the check dam at A3 can also be filled with release water from the newly implemented check dams upstream. The higher crest level in scenario 2 causes higher groundwater heads in that scenario at this location during that period.

At location A4, the situation is identical to A3; a check dam has been implemented in scenario 1 and the dam level has been raised in scenario 2, leading to the highest groundwater heads in scenario 2. A lag was identified between scenario 1 and 2 due to the retention effect of the upstream check dams in scenario 2. This can also be seen in the additionally displayed water levels directly at the check dam (Rajaveni *et al.*, *submitted*).

In summary, the results indicate that additional check dams have a positive (local) effect on the groundwater heads, just as the raising of the dams, though the effect is considerably smaller. The results also show that the implementation of additional check dams can retain water further upstream, possibly leading to a delay or even a lack of groundwater recharge in the downstream part of the catchment.

Conclusions and outlook

An integrated surface and groundwater model using MIKE11 and FEFLOW has been setup and was successfully calibrated. With the model it was possible to display salt water intrusion processes. Using the scenarios presented in this chapter, which show a significant local effect of the MAR structures on groundwater levels, the model is ready to be used to analyse the benefits of MAR structures on the saltwater intrusion process. For this, long-term analyses will be necessary. These simulations could be set up using yearly returning seasonal cycles for climatological conditions as well as natural groundwater recharge conditions. The simulations should cover a period of at least 50 years to analyse the effect of MAR structures on the long-term perspective. Furthermore, the model could also be used to predict the effect of long-term climatological changes.

14.3.2 MAR in a weathered crystalline hardrock aquifer: Maheshwaram, Telangana

Site description

One of the main experimental watersheds relevant for MAR studies in Saph Pani is located around the town of Maheshwaram (Figure 14.12) near Hyderabad, Telangana. With a total area of 53 km^2 and a semi-arid climate, it is situated on a weathered crystalline rock substratum, a geological and climatic context typical for the entire region where the saprolite weathering layer (10–20 m thick) is usually unsaturated. It is a watershed with a high density of groundwater production wells (>700) mostly for paddy irrigation. Changes in land use have occurred since 2006, the new Hyderabad international airport being located less than 10 km away. It is expected to become a peri-urban area in the coming years as significant housing projects are planned. MAR systems exist throughout the watershed in the form of percolation tanks, check dams, defunct dug wells, etc.

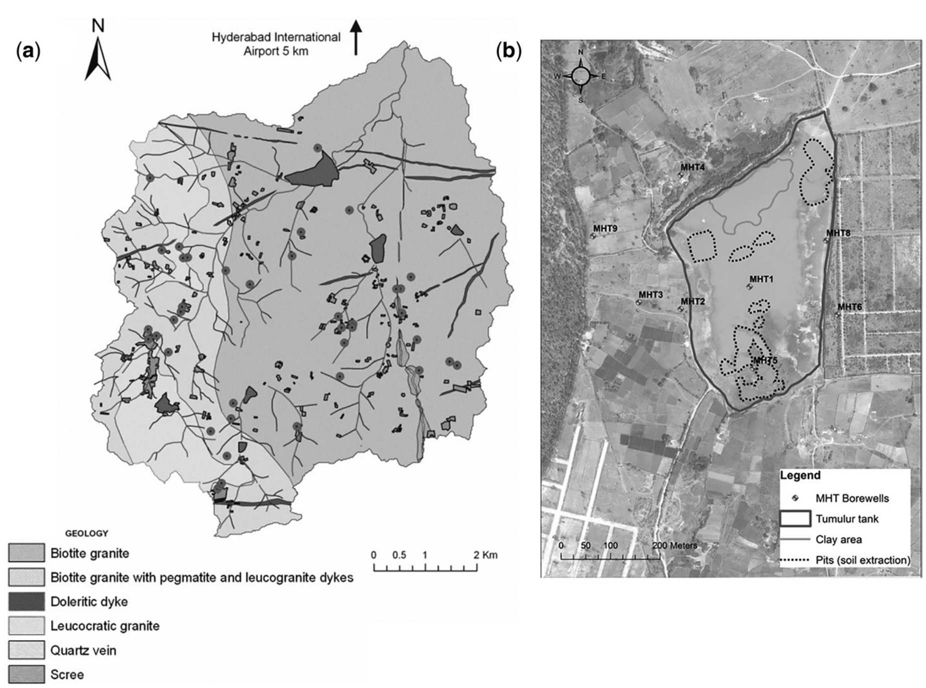

Figure 14.12 (a) Geological map of Maheshwaram watershed. Main MAR (percolation tanks and defunct dug wells) structures are indicated in dark grey. Middle grey areas are irrigated paddy fields. (b) Tumulur tank structure with pumping and observation wells (open circles).

Intensive groundwater exploitation for irrigation has resulted in aquifer over-exploitation and deterioration of groundwater quality (fluoride above maximum permissible limit of 1.2 mg/L (BIS, 10500:2012), salinisation and agricultural inputs). MAR is an attractive concept for groundwater augmentation and enhanced groundwater quality near wells exploited for domestic use.

Problems to be solved

The modelling objective for the case study in Maheshwaram is triple:

1) Develop tools to take into account the highly variable geometry of percolation tanks on weathered crystalline basement rocks under the specific Indian climate (dry season vs. wet season) through a specifically developed module for the 3D finite difference transient groundwater flow model MARTHE (Thiéry, 2010).
2) To assess the influence of percolation tanks on water quantity at local and regional scale.
3) To assess the influence of percolation tanks on crystalline basement rocks on water quality, in particular on fluoride concentrations, triggered by water-rock interactions with fluoride-containing minerals and evaporation together with agricultural backflow on paddy fields.

Tools and modelling strategy

Modelling infiltration from percolation tanks of variable geometry via a partially saturated weathering zone To assess the performance of percolation tanks the three-dimensional, finite difference transient state numerical groundwater code MARTHE was optimized by implementing three-dimensional, non-perennial surface water bodies in continuity with groundwater via an unsaturated zone. Implementation included the spatiotemporal evolution of the natural percolation tanks (i.e. changes in volume and geometry) linked to topography, taking into account heavy rainfalls during monsoon, evapotranspiration, infiltration, runoff, and groundwater dynamics. Part of the rain water stored in such tanks during the monsoon season infiltrates into the soil (variably-saturated media) and reaches the aquifer, while the rest evaporates. Theoretical simulations show that the new developed module "LAC" is able to simulate the relation between surface water and groundwater while respecting the water balance and to assess the highly variable geometry of infiltration tanks over the dry and wet season (Figure 14.13).

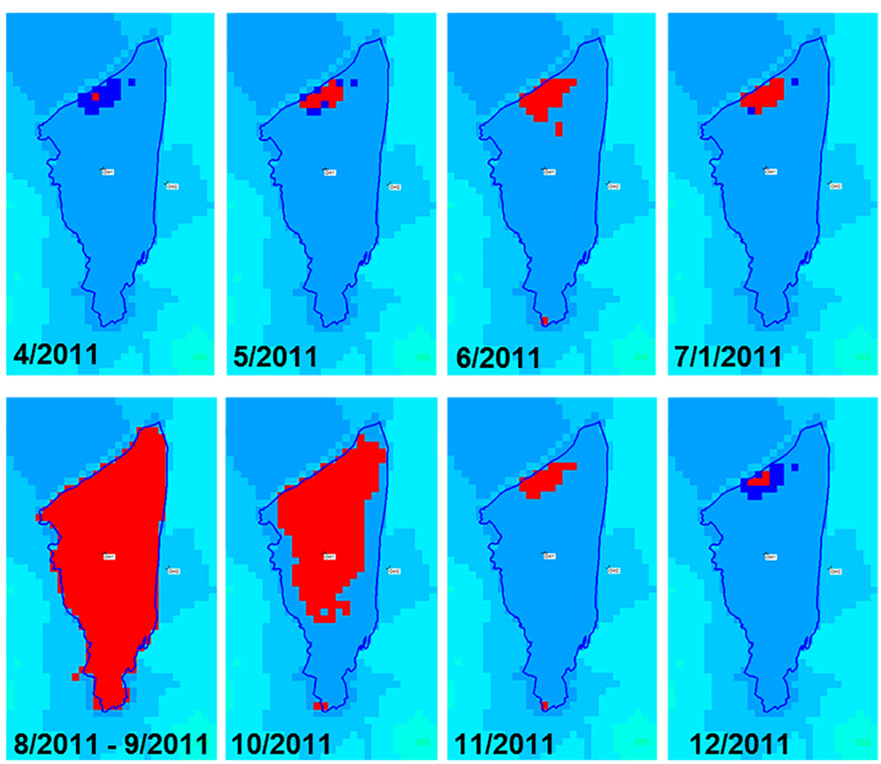

Figure 14.13 Simulated variable extension of the Tumulur tank filling (dark grey) over a monsoon season in 2011, Maheshwaram study site near Hyderabad, Telangana, India.

Modelling influence of percolation tank systems on fluoride concentrations: A geochemical model of solute recycling had been developed previously (Pettenati *et al.*, 2013) for paddy field irrigation using a 1D PHREEQC reactive-transport column (Parkhurst & Appello, 1999). This model was further developed and adapted to the percolation tank problem, on the basis of new monitoring data in order to test the conceptual geochemical model of MAR (Figure 14.14).

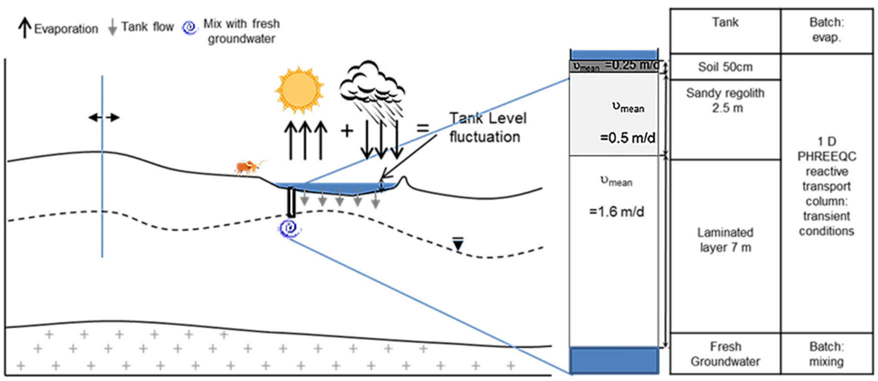

Figure 14.14 Conceptual model of hard-rock aquifer in southern India with Managed Aquifer Recharge (MAR) through an infiltration tank used for the development of a1D Phreeqc reactive column model. ν_{mean} is the mean pore flow velocity (Pettenati *et al.*, 2014).

Reactive transport column modelling was performed over a period of 110 days with the calculated pore flow velocity, taking into account the mineral composition of the 3 distinct layers of altered biotite granite (determined by XRD analysis) the Cation Exchange Capacity (CEC) of the weathering profile (determined by cobalt hexamine chloride solution) and the measured initial groundwater composition.

Outcome, added value and perspectives

The 3D MARTHE software was first developed in 1990, and already integrated surface-groundwater flow under varying saturation states including density driven flow (Thiéry, 2010). It is now ready, with the implementation of a specific module for percolation tanks, to be applied to MAR systems on weathered crystalline basement rocks in India and elsewhere. Such massively integrated models are still the exception and will be increasingly used as decision-making tools for assessing the quantitative effects of MAR on groundwater resources at the watershed scale.

The geochemical 1D reactive transport model, using PHREEQC, investigated the role of managed aquifer recharge under variable climatic conditions and its impact on groundwater chemistry. A previous model satisfactorily reproduced the solute behaviour in Maheshwaram groundwater under the influence of paddy fields (Pettenati *et al.*, 2013). Based on that model, the reactive transport model of the Tumulur tank infiltration through the critical zone helps to understand the evolution of fluoride enrichment or depletion in groundwater when MAR is implemented in a watershed. Results of the first scenarios simulation show that the beneficial effect of MAR may be variable over the year, being strongest during monsoon where significant dilution occurs, whereas during the dry period F- accumulation occurs. In sum, the beneficial effects observed during monsoon are countered by the adverse effects during the dry period so that no overall water quality improvement related to the MAR system can be expected at neither the local nor, most likely, at the regional scale. Extrapolating to the regional scale would require integration of 3D groundwater flow approaches with the developed geochemical model.

14.4 MODELLING OF WETLANDS

Natural wetlands play an important role in regulating surface water and groundwater flows within a watershed and also possess a purifying quality through intensive and diverse biological processes, ranging from macrophyte uptake of nutrients and contaminants to microbiological processes. Those processes are voluntarily used and optimised in constructed wetlands. In an intermediate position between natural and engineered systems, there are man-made wetlands used for agriculture, notably paddy fields, with important effects through (1) supplementary water abstraction from the watershed, both from surface water and groundwater, (2) enhanced evaporation, (3) nutrient and trace element uptake by crops and (4) agricultural return flow towards the aquifer. Effects on groundwater quality may be either beneficial (through filtration, water-rock interaction, biological processes in soil, the underlying variably saturated zone and the aquifer, in an analogous way to SAT systems) or, on the contrary, adverse (mainly through evaporation and return flow causing enhanced salinity, trace elements and wastewater-related contaminants and pathogens). In this sub-chapter, we investigate the impact of indirect wastewater recycling for irrigation in a peri-urban watershed in Telangana through an integrated modelling approach.

14.4.1 Integrated modelling of the Musi River Wetlands: Hyderabad, Telangana

Site description

The Musi River is a major tributary of Krishna River, originates in the north west of Hyderabad in Rangareddy district and flows down in a south east direction, passes through Hyderabad city and then joins Krishna River at Wazirabad in Nalgonda District. The Musi River has been intercepted by two major reservoirs, Himayat sagar and Osman sagar, upstream of the city. Below these two reservoirs, the Musi River receives only the city's wastewater and storm water. It receives water from its upper catchment only when excess flood water is released from these reservoirs. The study area lies between coordinates 17° 15′ N, 17° 30′N and 78° 30′E, 78° 45.0′E and includes the villages Peerzadaguda, Kachiwani singram and Mutialguda, situated in peri-urban Hyderabad and on the northern side of the Musi River (Figure 14.15).

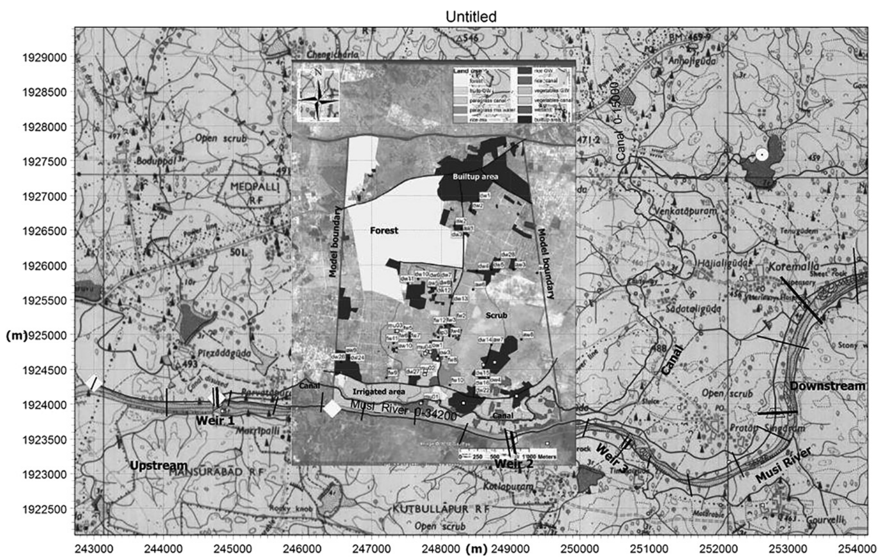

Figure 14.15 The study area east of Hyderabad, showing the Musi River, and the irrigation canal.

The Musi River downstream of Hyderabad has a cascade of overflowing weirs/ponds at which water is diverted into irrigation channels on both sides of the river. The wastewater flow in the river has made it a perennial river which is a significant resource in this semi-arid peri-urban environment where the cultivation of fodder grass, paddy and vegetables has provided economic benefits to many peri-urban inhabitants. Year round cultivation, water in the irrigation canal, overflowing diversion structures (weirs), storage ponds etc. have resulted in arise of the water table and converted the riparian zone along the river into a wetland.

Problems to be solved

Wastewater irrigation vs natural treatment systems Wastewater reuse has become a major area of interest to engineers, biologists, chemists, agronomists, water supply authorities, industries, water resources authorities, etc. Different agencies and stakeholders have different concerns such as prevention of surface water pollution, conservation and recycling of soil nutrients and development of additional water sources for agriculture, industries or non-potable supplies. Irrigation practice with wastewater is one of the reuse options for wastewater. The livelihood and economic activities of peri-urban farmers are the key drivers of wastewater reuse, especially irrigation for agriculture production and, as a secondary advantage, by-products and indirect benefits, for example, cheap water, perennial supply, reduction in surface water pollution, increase in soil nutrient and groundwater recharge (increase in specific yield of underlying aquifer increases). Some obvious downside aspects are soil degradation, degradation of ambient groundwater, cropping pattern change, aesthetics, and health risk for consumers and farmers.

Wetlands "Wetlands are areas where water plays an important role, creating a suitable environment for the associated plant and animal life. They occur where the water table is at or near the surface of the land, or where the land is covered by shallow water" (Ramsar, 2013). Wetlands, whether human made, (i.e. constructed) or purely natural, are also considered to be a cheaper and low-cost alternative technology for wastewater natural treatment. Distribution and differences in the types of natural wetlands are caused by topography, soil, drainage, vegetation, geology, climate, land use, as well as infrastructures like canals, controlled and impounded natural drainages or human-induced disturbances. Water table depth and its temporal variability and movement of water from one level to another through the wetland are the key parameters in characterizing the types and behaviour of a wetland.

Groundwater surface water interactions Surface water and groundwater have often been managed separately by completely different branches of the government. It is now recognized that water resources problems cannot be treated in isolation. Problems like wetland protection or the conjunctive use of surface water and groundwater resources require the integrated management of surface water and groundwater including water chemistry and ecology. Increasingly, water resources are managed on a watershed basis while addressing problems at the local scale. Watershed-based water management systems require new and more sophisticated tools. Traditional groundwater and surface water models were not designed to answer questions related to conjunctive use of groundwater and surface water, water quality impacts of surface water on groundwater, impact of land-use changes and urban development on water resources, and floodplain and wetland management. Instead, fully integrated hydrologic models of the watershed behaviour are required.

Objectives The main objective of the study was to understand the hydrodynamic behaviour of the groundwater-surface water systems under the influence of anthropogenic activities like irrigation, canal construction (seepage), weirs/ponding in the natural drainage of a riverine wastewater impacted (agriculture) wetland. The better understanding of the surface and sub-surface hydrologic processes in an integrated manner will help in assessing the positive and negatives impacts of wastewater irrigation practice on the groundwater and surface water systems. Considering the overall objective of the study dealing with models, it was necessary to understand the movement and exchange of water among the various zones of the system like the overland surface, unsaturated zone (sub surface), aquifer, vegetation and the exchange with surface water bodies (rivers/canals).Therefore, a distributed hydrologic tool, MIKE SHE was selected for carrying out the study.

Tools and modelling strategy

Integrated catchment modelling: application of MIKE SHE

MIKE SHE has been widely used for integrated hydrologic modeling. MIKE SHE's process based framework allows each hydrologic process to be represented according to the problem needs at different spatial and temporal scales. The water movement module of the software has a modular structure which includes six process-oriented components of the hydrological cycle. These are interception/evapotranspiration, overland/channel flow, unsaturated zone, saturated zone, snow melt and the exchange between aquifers and rivers (Figure 14.16) (DHI, 2014). MIKE SHE uses MIKE 11 to simulate channel flow and interact with surface water. MIKE SHE 's strength lies in its feature to provide a simulation of coupled, unsaturated-saturated zones, interaction between evaporation and shallow water tables and a better evapotranspiration module with root zone exchange apart from efficient coupling with open channels.

Modelling strategy In the present case, the area of interest was wastewater irrigated area along the Musi River which includes the river, weirs, cultivation practices, pumping etc. However, in this instance also, it appeared that the model domain needed to be suitably up-scaled to have a realistic groundwater boundary. For this reason, the model catchment was up scaled towards upland on the northern side up to near village Narapalli (Figure 14.16). The main input parameters for the model setup include topography, soils, land use and land cover, natural and canal drainage networks, locations of weirs and their hydraulic parameters, well numbers and locations, agriculture and irrigation data, rainfall, potential evapotranspiration, aquifer parameters etc. These parameters were gathered from field visits, primary survey, monitoring and also taken from secondary sources of research reports conducted in the area. The model domain (12.68 km^2) was divided into 60 m x 60 m cells. The irrigated area inside the model domain is about 1.73 km^2. In the present model, the study area is very small and highly vegetative and in the catchment no stream or ditch of significant size which carries significant surface runoff during dry or even rainfall period is present. However, there was a good number of observed groundwater table data across the model domain; hence the model was calibrated with groundwater depth only. All the processes like overland flow, unsaturated zone flow, saturated zone, evapotranspiration and exchange with surface water were included in the model setup as well as simulations considering their roles in the wastewater irrigation practice as a natural treatment system, i.e., soil-aquifer-treatment. MIKE 11 was setup and simulated as stand-alone including the Musi River, canal and Weir 2 and later on was integrated/ coupled with MIKE SHE. The coupled length of the Musi and the canal with MIKE SHE are 2.28 km and 4.05 km respectively.

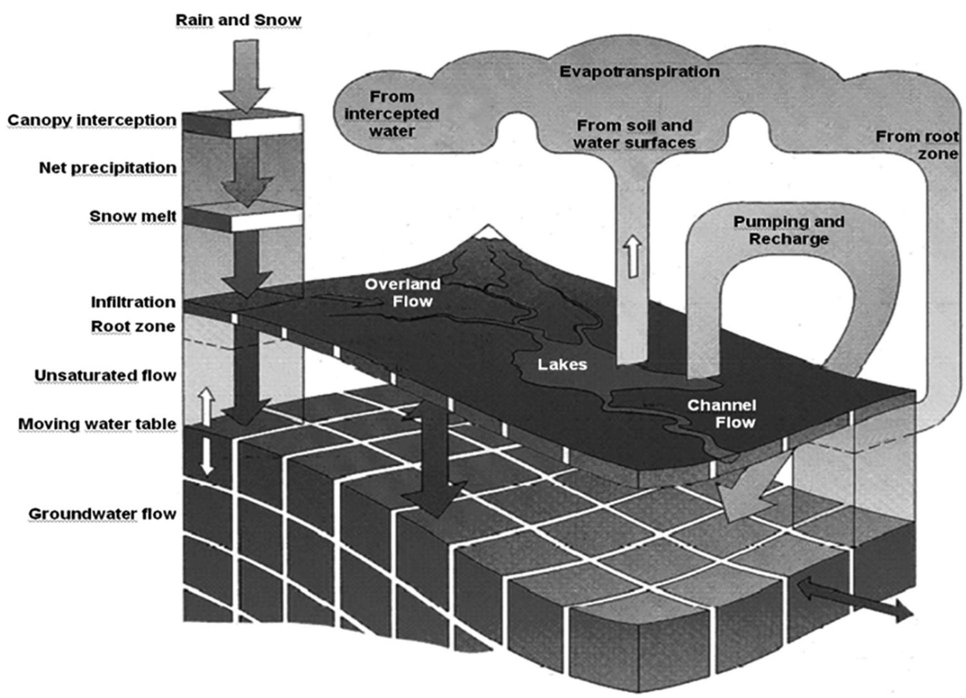

Figure 14.16 Hydrological processes in MIKE SHE (DHI, 2014).

Outcome, added value and perspectives

Two basic scenarios were simulated. The pristine scenario, assuming there are no canals and weirs/ponds across the Musi River and no additional irrigation except rainfall, and the baseline scenario, i.e. the existing condition. Irrigation, weirs and canal seepage have changed the hydrodynamic characteristic of the area; it is functioning like a wetland where there is direct exchange of groundwater with overland flow and the water table is close to the surface. The results shows that the area with a groundwater table within 6 m from the surface, i.e. Wetland, (Ramsar, 2013) has increased from 0.74 km^2 (Pristine condition) to 1.47 km^2 (Baseline) over the years (Figure 14.17).

In order to evaluate wastewater irrigation practices through the purifying action of agricultural return flow through soil, the variably saturated zone above the groundwater level and the aquifer itself as a natural treatment system, the first requirement is to know the movement of water through various zones and to quantify the exchange of water among them through water balance analysis. The MIKE SHE water balance tool provides a detailed account of the water balance. The water balance in terms of mean annual flow (Million cubic meters, Mm^3) including the losses and the return flows from different components of the system is presented in Table 14.3.

Overall the groundwater flow gradient is towards the Musi River and the gradient inverses locally due to pumping. Overland zone and saturated zone are interacting and exchanging water directly which is a typical feature of a wetland. In addition to salinity due to wastewater application salinity occurs because of soil and saturated zone evaporation. Farmers apply water when the deficit reaches around 50% (Maximum allowable deficit = 0.5). Even though wastewater supply is continuous and free, and farmers are conscious of the benefits related to the free nutrients wastewater contains, water application in the area is limited mainly by two factors, (1) the energy needed to lift water from the canal to the upland area and interruptions of power supply and (2) the farmers' fear of unnecessary contact with poor quality water. Thus over-irrigation and pumping occur, especially in the paragrass and vegetable growing areas. In the irrigated area, the consumptive loss is about 25% of total inflow and the return flow is therefore 75%. The modelling confirms the infiltration from Musi to the aquifer in the upstream of the weir where the ponding level is above the groundwater level. Seepage from the canal contributes to a rising water table and return flows. The stretch just downstream of the weir receives water from the aquifer (base flow) and is in gaining state.

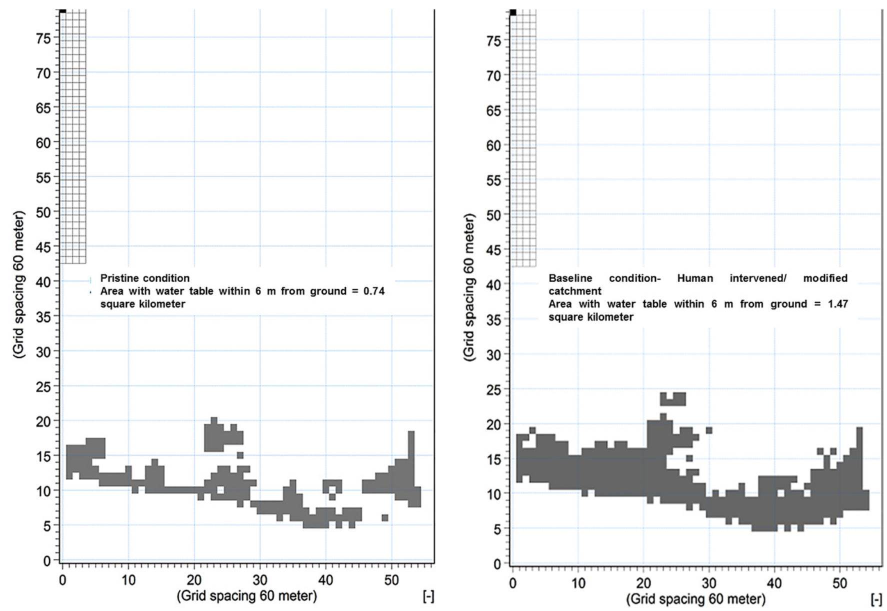

Figure 14.17 Impact of irrigation, canal and weirs on water table: **a.** pristine condition and **b.** baseline condition.

Table 14.3 Detailed mean annual inflows, losses and return flows in million m^3.

Components	**Pristine (model Domain –12.68 km^2)**	**Baseline (Model Domain –12.68 km^2)**	**Baseline (Irrigated Area – 1.73 km^2)**
Inflow			
Precipitation	10.22	10.22	2.14
Irrigation	0.00	10.29	10.17
Infiltration from canal & Musi river	0.00	2.57	2.56
Total	10.22	23.08	14.87
Loss			
Canopy evaporation	0.55	0.62	0.15
Overland evaporation	2.44	5.26	2.95
Soil evaporation	0.71	0.64	0.13
Saturated zone evaporation	0.02	0.03	0.03
Plant transpiration	2.40	2.11	0.48
Total	6.12	8.67	3.73
Return flow			
Direct surface return flow	1.56	7.35	5.73
Sub surface(interflow)	0.16	5.11	4.27
Base flow-through aquifer	1.40	0.92	0.95
Groundwater recovery	0.97	0.97	0.17
Storage	0.00	0.04	–0.01
Total	4.09	14.39	11.10
Error	–0.01	–0.02	–0.03

In addition to other parameters like soil, geology, vegetation, irrigation practice etc. the key system parameters for wastewater irrigation and for the natural treatment related to agricultural return flows through different milieus (soil, variably saturated zone and the groundwater body), are suitable topography and boundary conditions. Human interference in terms of irrigation infrastructures has increased the area of the riverine wetland. Canal seepage and ponding at weirs have made a significant contribution to the rise in water tables. Wastewater application on land has increased salinity (Biggs & Jiang, 2009; Hofstedt, 2005; McCartney *et al*., 2008; Ensink *et al*., 2009) but high water tables might also have contributed to soil salinity. However, the positive outcome is that the specific capacities of wells in and around the irrigated area have increased. To protect native groundwater resources outside the wastewater irrigated system (soil aquifer treatment system), the movement of wastewater in the wastewater irrigated area could be managed with well-planned recovery wells (appropriate locations, capacity, types and depths pumping schedules, etc.) and artificial or natural collector drains (appropriate depth, size, locations etc.) Therefore the share of groundwater and (wastewater-containing) surface water used for irrigation should be optimised through watershed–wide, integrated modelling to maximize the benefit and minimize the negative impact of wastewater irrigation practices. Several important agencies need to play an active role in encouraging and regulating wastewater irrigation practice in this area for example the Department of Irrigation and Agriculture and The State Pollution Control Board.

The distributed hydrologic modelling of the Musi wetland using MIKE SHE has demonstrated MIKE SHE's ability to represent complex hydrological systems found within many wetland environments where groundwater and surface water interactions are common hydrological processes. The detailed water balance analysis helps us to understand the movement and quantity of water from one level to other.

14.5 GENERAL CONCLUSIONS

In the light of the modelling exercises applied to the different NTS case studies, the biggest challenge for modelling NTS is model integration. When looking at NTS's like constructed wetlands or percolation tanks (soil-aquifer treatment) we need to take into account surface runoff, the unsaturated soil zone (complex but crucial for water purification), the saturated groundwater flow and even the density-driven saltwater flow in coastal aquifers. Water flow is a continuum but most currently available models are not yet able to treat it as such. One of the major advances in Saph Pani was the establishment of integrated models that take into account the whole water cycle at the watershed scale from surface flows, unsaturated and saturated flows to density driven flows. The project studies also integrated scales: Modelling NTS's needs both, a close look on their behaviour at a very local scale but also upscaling to a watershed scale to simulate effects if a large number of them were implemented. A typical example is percolation tanks. Our observations at the Maheshwaram site showed that their extension in all three dimensions varies widely with rainfall from close-to-nil during the dry season to maximum extension during monsoon. Treating their geometry as constant over time is an oversimplification that can lead to erroneous results if we want to estimate their real impact on groundwater recharge. For this reason, a specific module simulating infiltration was developed for the MARTHE software, already massively integrated with respect to all flow types (surface flow, unsaturated, saturated and density driven flow), able to simulate realistically the behaviour of infiltration tanks from rainfall, evaporation data and surface topography.

Another type of integration that revealed to be a crucial factor was the effects of water flow on water quality changes. Here, the most instructive example from the Saph Pani project is the simulation of ammonium transport from the heavily polluted Yamuna River, across the alluvial aquifer before reaching the wells that pump the river's bank filtrate. Ammonium breakthrough was first measured and modelled at the laboratory scale through percolation experiments in sediment columns and then up scaled to the aquifer scale through reactive transport modelling. An important result is the considerable residence time of several decades of ammonium in the aquifer due to sorption onto the aquifer material.

Models have been developed for all three types of NTS's studied in Saph Pani; managed aquifer recharge combined with soil-aquifer treatment, constructed wetlands and river bank filtration. This has demonstrated how these approaches can be used for understanding, planning and optimising NTSs. The modelling tools used are widespread and accessible (e.g. MODFLOW, FEFLOW, MIKE11, MARTHE, and MIKE-SHE). Even though the application of those tools to the specific problems of NTS implementation in the Indian context requires specialists trained in integrated modelling of complex systems like NTS's on different scales (up to basin scale), the knowledge and knowhow created in the project needs to be transmitted widely to young scientists and engineers through training programmes organised by the Indian institutions that were involved in the development of those methods within Saph Pani.

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