Chapter 11

Characterization and performance assessment of natural treatment systems in a Wastewater Irrigated Micro-watershed: Musi River case study

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11.1 INTRODUCTION

In India, many agricultural landscapes close to the cities have been irrigated with wastewater for many years. Fresh water scarcity, easy access, ready markets for high value crops have been the main reasons for its common reuse practice (Scott *et al.*, 2004; Qadir *et al.*, 2010). The water demand can only become worse in the future, with demands exceeding the supply where more than 40% of the population live. It is estimated that in around 12 years, nearly 60% of the world's population might face scarcity of water. Being an agriculture-based economy, India's irrigation water demand is the highest compared to all the other sectors, and given the increasing demand for fresh water, there is an urgent need to concentrate on efficient water resource management through enhanced water use efficiency and safe use of wastewater in agriculture (WHO, 2006). Wastewater reuse can be seen in diverse settings, and therefore, context specific. For example, when used for agriculture, the water can be simply diverted from storm water drains, or lifted from polluted rivers. Thus, the quality of this source water can vary significantly and requires analysis and treatment prior to safe reuse in agriculture (WHO, 2006).

Wastewater can be treated by land application, under natural conditions, and there are also other Natural Treatment Systems (NTS), like sedimentation ponds and wetlands that are effective (Arceivala & Asolekar, 2006). Many types of NTS that are effective in the treatment of wastewater have been described in the literature (Reed *et al.*, 1995; Shipin *et al.*, 2005; Crites *et al.*, 2014). The low-cost nature of the systems has been particularly appealing to those that find the traditional sewer network systems expensive. Where the soils and the groundwater conditions are appropriate, treatment of wastewater has been achieved for artificial recharge of groundwater, and the infiltration through the soils have facilitated the upgrading of the quality of water (Pescod, 1992). Thus, the soils and aquifers can treat wastewater to improve the quality, and where continuous wastewater irrigation takes place, the action of land application of wastewater can have a treatment effect. In conventional terms this is referred to as soil aquifer treatment (SAT). Naturally occurring wetlands are also effective in treating wastewater, but sometimes poorly investigated.

A number of previous studies on the Musi River catchment have documented the landscape dynamics (Mahesh *et al.*, 2015), hydrogeology and water quality (Perrin *et al.*, 2011a; Schmitt, 2010; Amerasinghe *et al.*, 2009), socio-economics (Buechler *et al.*, 2002), health impacts of wastewater use (Srinivasan & Reddy, 2009; Wakode *et al.*, 2014), soil and salinity implications (Biggs & Jiang, 2009), and willingness to pay for cleaner water (Mekala *et al.*, 2009). However, no studies have been undertaken to examine the natural treatment potential of the site where long term wastewater irrigation has been practiced. In this study, we attempted to characterise and assess the treatment performance of NTS in the micro-watershed.

11.2 STUDY SITE

The Musi River is part of the Krishna River basin, and is associated with an ancient irrigation system comprising a large wetland system. The river starts from the Ananthagiri hills and feeds the Krishna River after passing through the city of Hyderabad (6.8 million population, Census, 2011), picking up over 1.2 million m^3/d of wastewater (both domestic and industrial) from the city, which is a mixture of partially treated or untreated water (NRCD, 2001; Van Rooijen *et al.*, 2005: Amerasinghe *et al.*, 2012; Starkl *et al.*, 2013). The wastewater is used downstream for irrigation, either directly via a system of irrigation canals or after storage in tanks. The wastewater is a significant resource in this semi-arid peri-urban environment where the cultivation of fodder grass, paddy and vegetables provide economic benefits to many poor inhabitants of the area (Jacobi, 2009). Year round cultivation, which generates large return flows from irrigated fields, also contribute to a large share of the aquifer recharge (Perrin *et al.*, 2011b). Within the micro-watershed shallow groundwater is also pumped for irrigation in areas where canal water is not accessible, or too polluted to irrigate certain crops according to farmers, especially rice.

The present study was carried out in a micro-watershed Kachiwani Singaram (KSMWS), comprising an area of 274 ha which is close to Hyderabad and is within the Musi River watershed (Figure 11.1). Along its 274 km length, 22 weirs provide temporary storage of water for a series of irrigation canals that run parallel to the river. The KSMWS receives the river water at the first weir, through an irrigation canal which runs approximately 15 km and ends in a village tank. Canal discharge rates were highly variable in time and space, and were generally low, i.e. less than 2 m^3/s. The micro-watershed's climate is semi-arid, with a mean annual rainfall of about 750 mm (most of the rainfall occurs between June to October) with both high spatial and temporal variability. Over the last 18 years, annual rainfall in Hyderabad varied from 535 to 1473 mm respectively for years 2000 and 2010. Most of the rain events occurred during the monsoon, which was from June to October. The mean annual temperature was about 26°C, although during the summer time the maximum temperature can reach up to 45°C. The Musi riverbed shows a flat topography (mean slope <1%) (Massuel *et al.*, 2007). The watershed geology consists of a basement made of orthogneissic granite also known as "pink granite" with granite, quartz and dolerite intrusions and showed a well-developed weathering profile. From a landscape view the area under study comprises agriculture land (irrigated with wastewater and groundwater), a small reed pond (wetland) in the middle, barren land and built-up areas in the northern region. The village Kachiwani Singaram after which the micro-watershed was named, is at the eastern border of the delineated area and is situated adjacent to the irrigation canal. The major crops grown in the area were rice, paragrass and vegetables. Both canal water and groundwater were used for cultivation, depending on the availability and access. Therefore, within the micro-watershed irrigation practices varied widely.

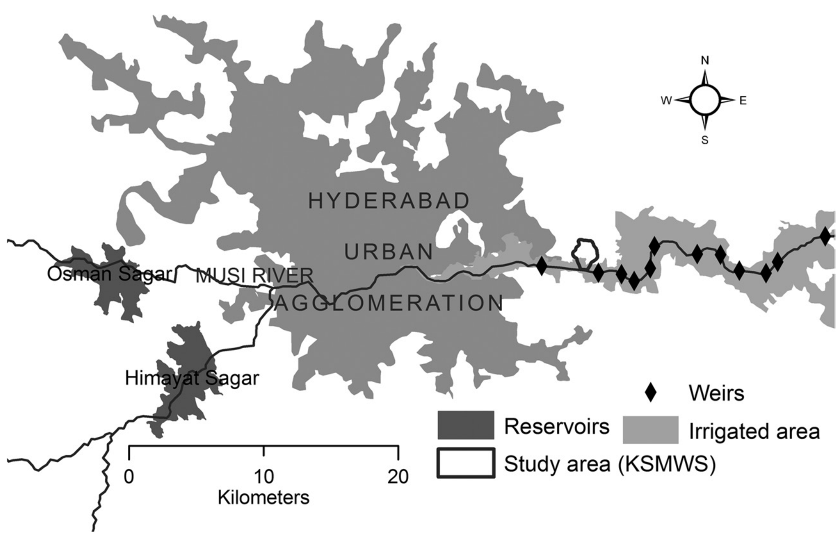

Figure 11.1 Location of the Kachiwani Singaram micro-watershed (KSMWS).

11.3 STUDY APPROACH

The micro-watershed was characterised using a number of activities, including land use surveys, discharge measurements, water level monitoring (continuous and piezometric campaigns), water budgets, pumping tests, geophysical surveys (electrical resistivity tomography), and water quality analysis (major ions, trace elements, biological oxygen demand, microbiology and pesticides). A performance assessment of the micro-watershed was carried out using past and current field studies spanning over 10 years. The flow and transport model for the local hard-rock aquifer system was developed to understand the treatment capacity within the micro-watershed.

11.4 MATERIALS AND METHODS

The methods followed for land use mapping, status of surface and groundwater, geophysical and hydrogeological characterization, and water quality assessments are given below.

Land use mapping was carried out using high resolution satellite imagery (1–12 m resolution, Google Earth images) and interpretations from a previous study (Amerasinghe *et al.*, 2009). Micro-level spatial land use variability was assessed using digital globe satellite data (dataset used: 21 January 2012) from Google Earth and validated with ground truth data, field observations and farmers interviews. Digital globe data covering the study area were used to investigate the spatial variability in cropping patterns and other land cover classes. The datasets were geo-referenced with UTM projection and WGS 84 datum. The datasets on cropping patterns, built-up areas, topographical maps, ground truth data and farmer interviews were used as inputs for classification and accuracy assessment.

River discharge measurements were carried out using a standard float method. A distance of 20 m, free of vegetation was selected for the measurements. If the flow was very slow, shorter distances were considered. Oranges were used as a floating object and a stopwatch was used to monitor the time taken to reach the specified distances. The experiment was repeated three times and the average was used for calculations.

Application of geophysical parameters such as electrical resistivity with geological perception was utilised to resolve the hydrogeological complexity of the near surface (Sonkamble *et al.*, 2013). Geophysical investigations such as electrical resistivity tomography (ERT) were carried out to delineate the subsurface lithological layers and saturated thickness. SYSCAL Junior Switch multi-node computer-controlled imaging system (IRIS make, France) was used with 48 electrodes connected to a multi-core cable. Wenner–Schlumberger and Dipole-Dipole configurations were selected to scan the subsurface profile of lengths ranging from 96 to 470 m (as per the availability of space) with a unit electrode spacing of 2.0 m to 10.0 m. A total of 17 ERT profiles were carried out at 11 different locations (Figure 11.2) to represent various stages of the weathering processes. A continuous ERT profile line A–A′ of 1.62 km distance was chosen to decipher the spatial variations of the regolith (saprolite) thickness and saturated zone, and also to determine the subsurface contamination from north to south orientation up to the Musi River. In addition to the A–A′ profile line, the ERT investigations were also performed at other locations in close proximity of observation wells. Surface modelling based on topography was also performed using ArcGIS software where the topography was estimated from the ASTER DEM with 30 m resolution datasets. Detailed information encompassing site geology, geomorphologic and hydrogeological conditions at each ERT profile was noted during the ERT investigations and was utilised in the geophysical interpretation of the images.

Monthly measurements of water levels were carried out in the four monitoring piezometers (W1, W5, W6 and W8) (Figure 11.3). The W1 was located south of the irrigation canal in the paddy field and captures a wastewater contaminated shallow unconfined aquifer. W5 was located 400 m north of the irrigation canal and located in a paragrass field irrigated with wastewater. W6 was at 700 m north from the irrigation canal, located in a paddy field irrigated with groundwater. Similarly, W8 was placed 1 km north of the irrigation canal, and positioned at an elevated topography, located in a vegetable farmland irrigated with groundwater. Automatic level-loggers (Solinst® Levelogger), recording water level and water temperature at a 30 min time interval were installed in W8 and W6. Four extended piezometric campaigns over the catchment were carried out in May, June (pre-monsoon), September (monsoon), and October (post-monsoon) of 2010. Through interpolation using inverse distance weighting, annual (pre-/post-monsoon 2010) and inter-annual comparison (post-monsoon 2010/post-monsoon 2013) maps were prepared and further analysed to highlight particular patterns of the groundwater movement.

Hydrodynamic properties were assessed by conducting hydraulic tests, i.e. pumping test, in the four piezometers (W1, W5, and W8). Drawdown was monitored during the entire pumping duration (recorded at three minute interval). Then the recovery was recorded every minute up to reaching the initial water level. These data were used to estimate transmissivity of the aquifer using the software WinIsape© developed by BRGM. The Theis method was used for interpretation (Theis, 1935), using a best-fitting procedure. Variations in water velocity inside the bore well during pumping were measured. For this, a flow meter was used with a propeller which turns when it comes into contact with the water flow (measure of flow velocity). Measurements were made at the bottom of the casing and extended to the base of the hole. Each productive fissure was indicated by a drop in flow velocity. Results were correlated with geological and observations of cuttings, to identify the productive zones.

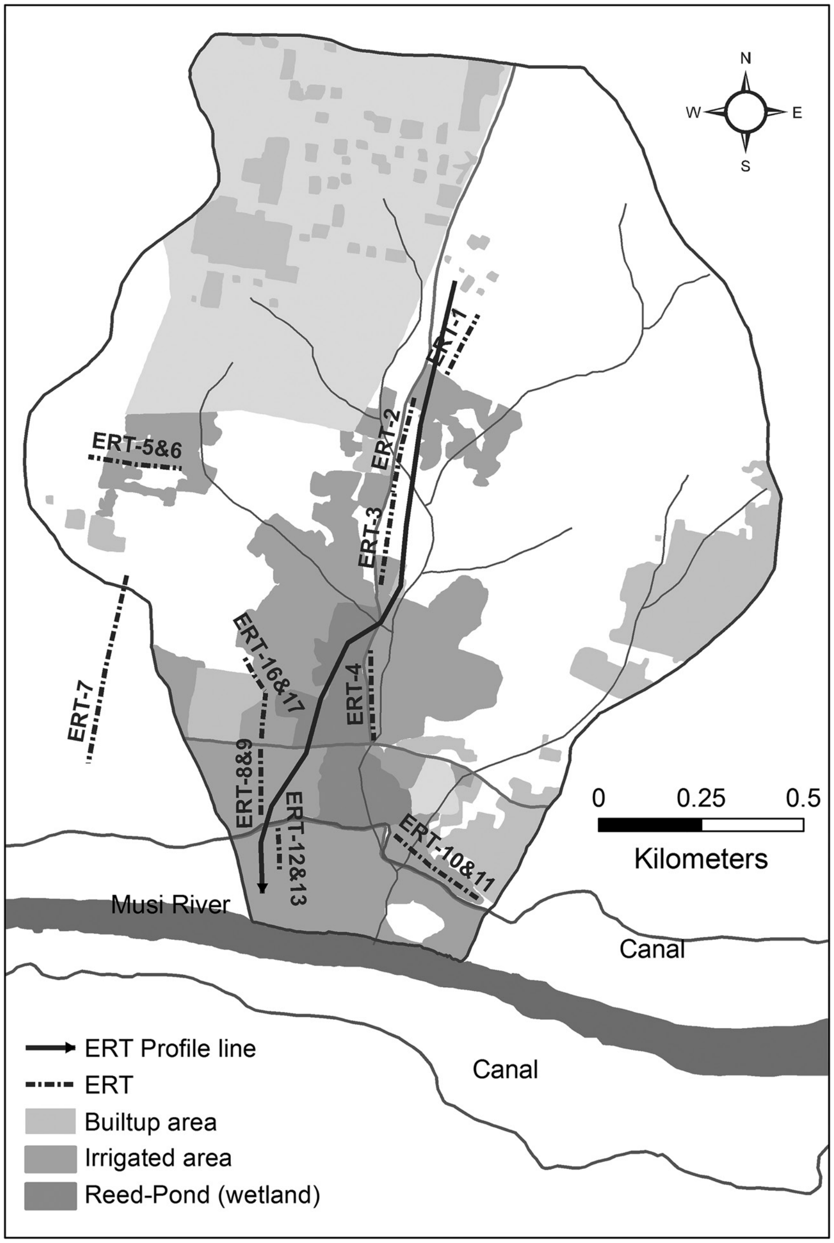

Figure 11.2 Geophysical measurement (ERT) locations in the KSMWS (N 17.390914. E 78.627036).

Two hydro-chemical water sampling campaigns were carried out in 2012 during the pre-monsoon (June) and post-monsoon seasons (November). Fourteen water samples were collected from canals and wells (Figure 11.3) and analysed for major anions and cations, heavy metals, selected microbiological parameters and pesticides. Eleven groundwater, 2 canal and a wetland water sample were analysed. The groundwater samples were from piezometers (4), bore wells (3 agriculture wells and 1 domestic well) and open dug wells (3). Groundwater and surface water samples were collected in 1 L polyethylene bottles for major cations and anions and before sampling the bottle was cleaned three times with sample water. Analysis was performed for major cations and anions as per the guidelines by APHA, 2005. The pH was measured using a pH meter; electrical conductivity using a conductivity meter; carbonates, bicarbonates, calcium and magnesium using a titration method; sulphate using a turbidity meter; fluoride using an ion meter with specific electrodes; sodium and potassium using a flame photometer and nitrate using an atomic absorption spectrophotometer. Pesticides samples were collected in 1 L amber coloured high density polyethylene bottles and kept below 4°C until analysis. Organochlorine, Organophosphorous and Carbamate pesticides were measured using a liquid-liquid extraction and method and analysed in a GC-MS (Gas chromatography – Mass

Spectroscopy) instrument. Results of previous sampling campaigns during the period 2006–2008 (Amerasinghe *et al.*, 2009; Perrin *et al.*, 2011a) were also used for comparison, and to understand the groundwater dynamics in the watershed along the Musi River as well as the impact of wastewater on groundwater quality.

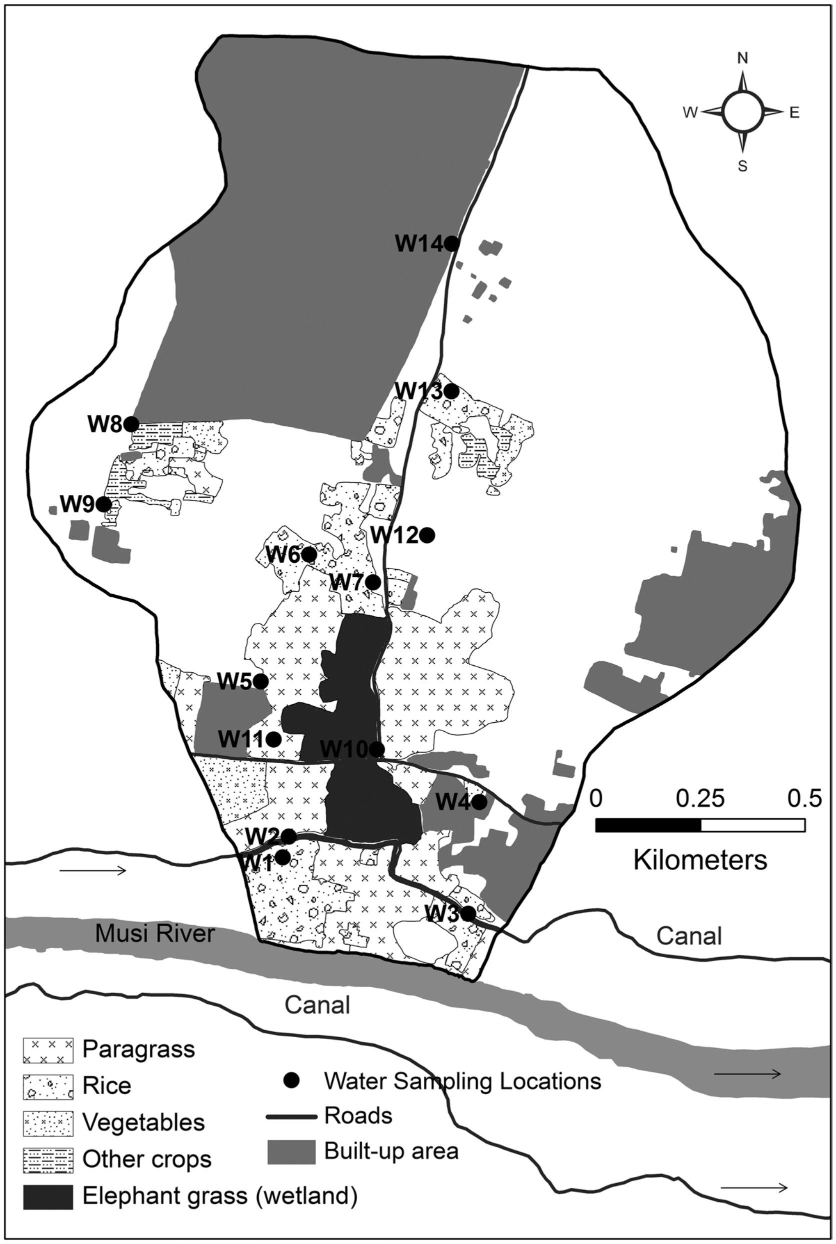

Figure 11.3 Land use classes and water sampling points in the KSMWS. W1-W14 are sampling wells.

11.5 RESULTS AND DISCUSSION

Characterization and performance assessment of the micro-watershed was based on a number of features, namely, land use, land cover, hydrogeology, geophysical attributes, water balance, aquifer characteristics and water quality. This agriculture-based micro-watershed depended on both wastewater and groundwater irrigation for its production systems. Closer to the canal the water levels in piezometers were high (up to 3 m below ground level), due to the continuous wastewater irrigation activities that occurred year round. In the north of the watershed, the water levels fluctuated over the seasons, due to high pumping rates of the irrigation and domestic wells during production times, and being the only source of water. Thus, across the micro-watershed, the performance appears to be strongly influenced by the wastewater irrigation practices, groundwater pumping and seasonal rainfall. Here, we discuss the SAT process and the performance of the natural wetland.

11.5.1 Land use, geomorphology, water balance and aquifer characteristics

The Musi River is a perennial river due to the urban wastewater discharges from the city of Hyderabad. Based on the current water supply, the wastewater generated from the city was estimated to be around 1.2 million m^3/d (Mahesh *et al*., 2015), which was channelled into irrigation canals, for agriculture use within the micro-watershed.

Flow rate measurement campaigns (river and canal) showed an increase in irrigation rates during the wet season (Figures 11.4 & 11.5). However, there was no clear correlation between the rainfall and river discharge because the influence of urban wastewater was far greater (Figure 11.4). Thus, at a local level the performance of the aquifer appears to be strongly influenced by wastewater irrigation, groundwater pumping and seasonal rainfall.

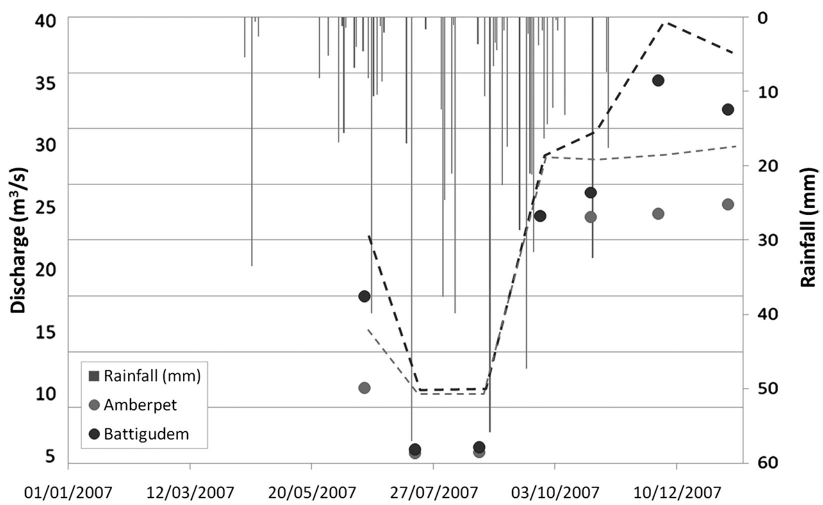

Figure 11.4 Discharge rate (m^3/s) measurements of the Musi River at Amberpet and Battigudem from May 2007 to January 2008 and daily rainfall (mm) in Hyderabad from January 2007 to January 2008.

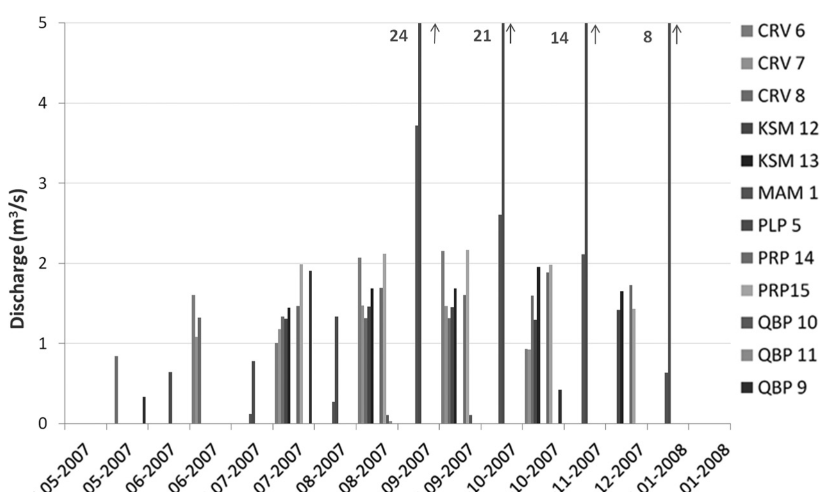

Figure 11.5 Discharge rate (m^3/s) measurements of the irrigation canals at 5 locations from May 2007 to January 2008. CRV = Chinnaravirala, KSM = Kachiwani Singaram, PLP = Pillaipalli, PRP = Parvathapuram, QBP = Quthbullapur are names of villages through which the canals pass.

A number of land use classes were identified and they were agriculture, areas under development, and barren lands. The irrigated area in the micro-watershed was approximately 48 ha and the major crops grown in the area were paragrass (56%), paddy rice (32%) and vegetables (8%). In terms of types of irrigation, 74% of the areas were under wastewater irrigation, and the remainder was groundwater. Paragrass was the dominant crop in the watershed, and wastewater was the main source of irrigation water. The micro-watershed showed visible signs of growth in infrastructure development and population, but the area under cultivation remained the same. The only difference was that as development processes engulfed the cultivated areas, new production sites were established (Mahesh *et al.*, 2015). We assumed that these changes would not affect the run-off or water balance and any increases in domestic use would be negligible compared to irrigation use.

As in many semi-arid environments, spatial and temporal interactions between surface water and groundwater were complex. The surface water percolation occurred mainly through preferential paths governed by topography and subsurface hydraulic properties. An analysis of the topography of the area using GIS tools indicated more than 10 small preferential flow areas within the micro-watershed. Localised recharge was also possible due to the uneven terrain. All these investigations revealed that the major surface run-off flow direction was towards the canal and then finally into the Musi River, which is in the north to south direction.

The hydro-geomorphology and the groundwater potential showed that the potential recharge conditions in the area varied from poor to good in the north to south direction of the micro-watershed. For example, the well yield in the northern part (pediment zone) was low, i.e. 30–40 m^3/d, and the recharge potential was also low, and restricted to the fissure zones. However, the recharge potential was high (up to 40%) in the shallow flood plain close to the Musi River (Figure 11.6).

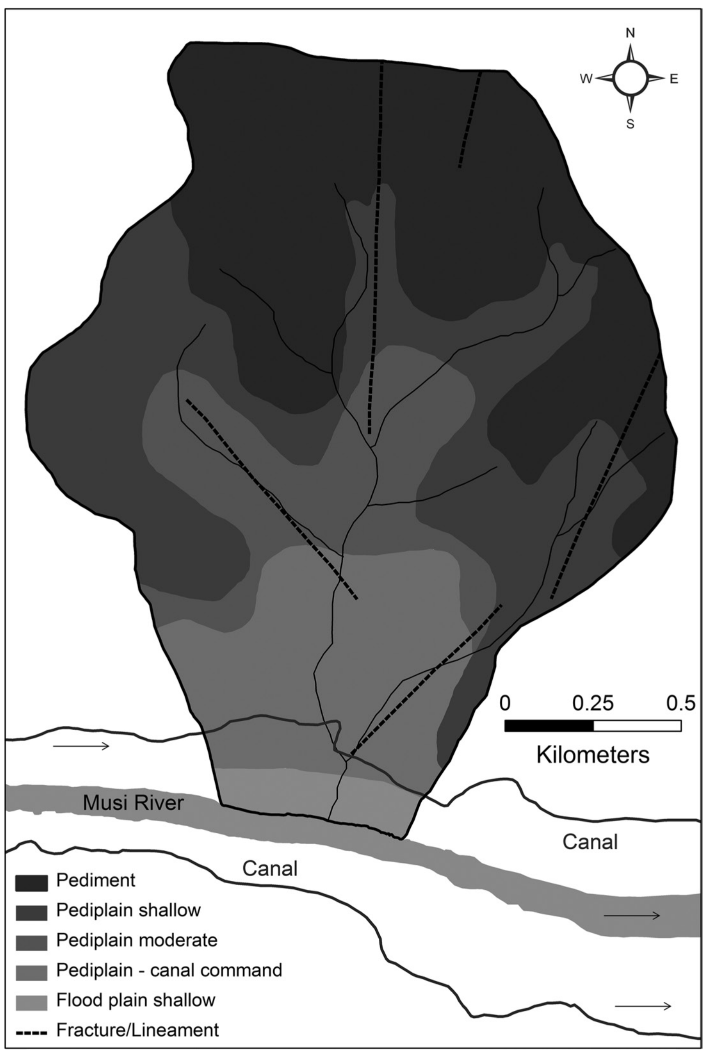

Figure 11.6 Geomorphology of the KSMWS.

The piezometric maps were significantly influenced by irrigation pumping during the pre-monsoon season (May and June). During the monsoon (October and November), the pumping influence was reduced, and consequently, a rise in the water table (5 meters) was observed in the central part of the study area. The high variability in the piezometric levels is understandable given the strong heterogeneities of the aquifer and the irrigation activities. Significant variations in the water level were captured daily, and even hourly, due to the periodic pumping action for irrigation. Piezometric levels (W1 and W5) in the southern part close to the canal water, did not vary significantly during the year and were within 2 m below ground surface. The water levels in this southern zone were controlled by the river which constitutes a boundary of the aquifer. The northern region piezometric levels (W6 and W8) were strongly influenced by the monsoon recharge that occurred from end of June as expected, in these unconfined crystalline aquifers with vertical recharge. The rapid rise of the water level after each rainfall event indicates the presence of rapid preferential flow paths in the saprolite zone due to the existence of preserved fractures (rise of groundwater levels less than one day after rainfall event). After the monsoon, the decrease of water level was fairly constant (dependant of natural flow and pumping) and between the years in those two bore wells.

When piezometric levels were compared between post-monsoon periods of 2010 (from a previous study) and 2013, two different zones of hydrologic activity could be delimited: A typical continuously decreasing water level due to over-exploitation of groundwater, especially during the dry season, and a gaining water level in the south influenced by abundant irrigation with wastewater.

The aquifer characteristics derived from the pumping tests of piezometers (W1, W5, W6 and W8) are given in Table 11.1. The pumping tests performed on four tube wells showed transmissivity values ranging from 9.9×10^{-4} to 3.4×10^{-3} m^2/s as expected in this geological context (Dewandel *et al*., 2006). Flow meter tests in the fractured zone showed that transmissivity values were controlled by a few productive fractures zones in each tube well (1 to 3) mostly on the upper part of the weathering profile. These results are consistent with previous studies (Dewandel *et al*., 2006; Maréchal *et al*., 2004) in the region showing that transmissivity was mainly constrained by the fracture zone at the contact between the poorly transmissive saprolite which ensure the storage capacity of the aquifer and the deeper fresher granite where only a few fractures may provide water in limited amount. The decrease of fracture density with depth may induce compartmentalisation of the aquifer in low water level conditions due to a decrease of connectivity (Guihéneuf *et al*., 2014).

Table 11.1 Aquifer characteristics of the KSMWS.

Piezometer	Discharge [L/s]	EC [mS/cm]	T [m^2/s]	Max Drawdown [m]	K [m/s]	S [–]
W1	0.63	1.4–1.55	9.9×10^{-4}	0.78	1.7×10^{-5}	3.02×10^{-4}
W5	0.61	1.55–1.63	1.1×10^{-3}	0.59	3.4×10^{-5}	3.03×10^{-4}
W8	0.55	0.95–1.15	3.4×10^{-3}	0.32	7.4×10^{-5}	8.90×10^{-8}
W6	0.18	1.65–1.7	–	2.35	–	

EC: Electrical Conductivity, T: Transmissivity, K: Hydraulic conductivity (K = T/e, with e, aquifer thickness), S: Storativity. Storativity values are indicative as tests were carried out in pumping wells (i.e., no observation wells) and during a short period.

11.5.2 Water quality

Surface water quality

The salinity and conductivity (EC) in the Musi River was high and showed an increasing trend downstream (McCartney *et al*., 2008; Amerasinghe *et al*., 2009; Biggs & Jiang, 2009). The EC in the canal water samples indicated an increase in the post-monsoon samples though not significant (1,217–1,490 µS/cm). This was contrary to the general expectation that monsoon would result in lowering the EC. In the wetland sample (W10) the EC was higher than in the canal reaching 1,750 µS/cm during post-monsoon period. The EC values in groundwater were higher than that for canal water. High temporal and vertical variability in EC logs were observed depending on the monsoon dilution or inversion of hydraulic gradient or placed close to the wastewater canal. Abrupt EC changes at around 30 m and 44 m below ground level in some wells suggested the presence of active flow fractures. Multiple factors may have contributed, amongst which weathering, silicate hydrolyses process and the hydraulic gradient may play an important role.

Major ions (sodium and chlorides), that contributed to salinity, did not vary significantly in the pre- and post-monsoon samples, however, values were indicative of anthropogenic influences. The wetland showed a greater variation in its constituents compared to the canal water. The canal water samples had variable nitrate concentrations ranging from 0 to greater than 150 mg/L. The high values could be due to the influence of agricultural activities as well as the source water, which carries urban run-off. Nitrate content in the Musi River and wetland (W10) were low (<20 mg/L) probably due to denitrification process in surface water linked to high organic matter loads (Reeds) (Lofton *et al*., 2007).

The total dissolved solids varied with time and decreased during the monsoon period, probably due to a combined effect of dilution from rainfall and run-off water. High concentrations of hydrogen carbonate, chloride, sodium and sulphate ions were common and with significant enrichments of sodium in the Musi River and chloride and nitrate in groundwater and an excess of magnesium for some samples of groundwater. In the surface water samples from the canal and wetland, the pH ranged from 7 to 8.3 and did not show much variation between the pre and post-monsoon samples

In 2007, the biological oxygen demand (BOD) levels in the Musi River exceeded 200 mg/L in the samples close to urban areas, but declined further downstream (around 40 km), except when some local activities like livestock bathing led to elevated levels. In comparison, for the same year, the BOD in the irrigation canal at the KSMWS ranged from 105–150 mg/L, over a period of 4 months (August to December). Apart from the biological contaminants of source water, livestock wallowing, and open defecation may have also contributed to the elevated levels. In the present study (2012), the BOD values decreased considerably, and ranged from 15–65 mg/L, probably associated with the improvements in sanitation infrastructure (Wastewater treatment plant rehabilitation and construction of two new ones) at the city level, during the past few years. Faecal coliforms were detected in the post-monsoon samples (>1,600 MPN/100 mL) however, *E. coli* O157:H7 was not detected. At present, local contamination may not be contributing to these values significantly. However, the hamlet of Kachiwani Singaram is expanding and will have an impact on the irrigation channels and therefore, there is a possibility that these canals may become wastewater drains, unless the wastewater disposal systems are set in place.

Groundwater quality

Groundwater quality showed a strong spatial variability in groundwater chemistry (i.e. mineralisation, long term wastewater irrigation, agriculture practices etc.). Two main poles of EC were visible: one representative of fresh groundwater with EC < 1,000 μS/cm, and one pole with groundwater influenced by canal water return flows with EC > 1,000 μS/cm. It is also clear that additional sources of groundwater contamination (e.g. agriculture, sewerage) existed with localised points showing quite high EC (even higher than 2,000 μS/cm). In the sector where groundwater is impacted by canal water return flows, groundwater EC was higher than raw canal water, most likely as a result of re-concentration by evapotranspiration processes and dissolution of ions by water-rock interactions. EC in post-monsoon 2012 is in general higher than the pre-monsoon samples in the southern part of the watershed. This may be due to the increase in hydrogen carbonate and calcium content or the strong influence of the wastewater from the canal.

Rainfall appeared to impact on the chloride, nitrate and sulphate concentrations. This may be due to the complex interactions with the local hard rock aquifer system. Except a slight increase in W1 post-monsoon samples, there was not much variation in the fluoride concentrations in groundwater. The concentration of nitrate and fluoride were above the permissible limits for drinking water (50 mg/L and 1.5 mg/L respectively according to the WHO guidelines, WHO, 2011). Sulphate reached values up to 411 mg/L (W4). It is evident that the groundwater chemistry with respect to each of the elements was influenced by long term wastewater irrigation, application of synthetic fertilizers, soil salinity, and ionic contributions from rock-water interactions and rainfall.

Increases in hydrogen carbonate ions in the post-monsoon groundwater samples were attributed to enhanced organic matter mineralization as a result of carbon dioxide, associated with the run-off processes. The high chloride content does not have a lithological origin in this hard rock terrain. It could be of anthropogenic and/or meteoric origin enhanced by evaporation processes in soils. The groundwater samples in the southern part of the catchment (W1 and W5) showed a sodium excess, indicating a strong cationic exchange process within the clay minerals in the soil. High contents of fluoride were due to rock-water interactions enhanced by irrigation return flows (Pettenati *et al*., 2013). Levels of nitrate and sulphate were attributed to agricultural practices and wastewater irrigation.

Hydro-chemical water facies showed that the temporal variations of major ions due to rainfall were not significant, however, anthropogenic activities like wastewater irrigation, application of fertiliser, soil salinity could contribute to high levels of nitrates and sulphates based on the level of activity. Excessive irrigation return flows, through solute recycling may have led to significant aquifer salinization further exacerbated by the presence of thick clay soils, which favour strong cationic exchanges in this area (Perrin *et al*., 2011a).

Of the 14 samples tested, only 7 samples were positive for pesticides. The pesticides fell into three families, namely, organochlorine, organophophorus and carbamate. Organochlorine pesticides like butachlor, organophosphorous pesticides like malathion and carbamate pesticides like carbofloro nuclon granules were used by farmers. While it is expected that the monsoon rains would dilute pesticide concentrations, in some samples there were increased levels after the monsoon, due to leaching and possibly agriculture run-off. Some pesticides like propoxure was 'not in use', but tested positive and cannot be fully explained. The permissible limit for drinking water as per BIS 10500 standards (Bureau of Indian Standards, 2012) for each pesticide is 0.01 μg/L, and those samples that were positive showed values higher than 0.01 μg/L. Even though people are not drinking the water either from the canal or local groundwater wells that were tested, the elevated concentrations of pesticide elements raise a question of ecological impacts and food production in the area. Faecal coliform range was higher in the post-monsoon samples (2–170 MPN/100 mL) than in the pre-monsoon samples (0.3–21 MPN/100 mL). A degree of soil remediation may have contributed to the reduction in levels.

Water budget

The water budget was expressed as follows (Perrin *et al.*, 2011a)

$$R + R_f + Q_{\text{in}} \pm L_c - P - E = Q_{\text{out}} \tag{11.1}$$

where R is the natural recharge during the monsoon, R_f is the return flows (mainly from wastewater/groundwater irrigation but also from domestic water uses), L_c is canal losses or gains, Q_{in} is groundwater inflow across groundwater reservoir limits, P is groundwater pumping, E is the evaporative discharge from the water table, Q_{out} is the groundwater contribution to Musi river base flow.

Then, the groundwater budget of the watershed can be expressed as follows: The final estimated groundwater outflow from the watershed or groundwater contribution towards Musi river base flow was 136 mm/yr. This means that the groundwater contribution towards the Musi River was 0.4% of the river flow in the study area. Thus, the wastewater irrigation contributed to the base flows of the Musi River significantly 70% of the irrigation return flow came from wastewater irrigation (113 mm/yr) and 30% was from groundwater extraction (29 mm/yr) and a small part of the return flow at the watershed scale (0.9 mm/yr) came from domestic water recharges (Perrin *et al.*, 2011a).

Natural wetland

Within the study site a natural wetland, comprising marsh grass, was also studied for its treatment potential. The wetland received return flows from the east and west sides where paragrass cultivation was carried out using canal water. A comparison of water quality between canal water (W2 = inlet) and the outlet point of the wetland (W10), showed that nitrate-N (W2 = 10 mg/L and W10 = 1 mg/L), sulphates (W2 = 305 mg/L and W10 = 68 mg/L), chemical oxygen demand (W2 = 32–248 mg/L and W10 = 16–144 mg/L) and BOD (W2 = 10–65 mg/L and W10 = 5–32 mg/L) were reduced during both seasons, indicating the natural potential for contaminant attenuation. EC was however high compared to all other samples and was attributed to the weathering within the wetland. The study indicated that the natural wetlands could have great potential for contaminant attenuation, and more detailed studies are required to understand hydro-geochemistry of this natural wetland.

11.6 CONCLUSION

Land use data, hydrodynamic monitoring, hydraulic tests, hydro-geophysical surveys, water chemistry data were useful to develop a conceptual model for flows and transport for the KSMWS, which consists of a hard rock aquifer (Figure 11.7). The study showed that there is a north-south gradient in the micro-watershed, which together with the different hydrological flows impacts the groundwater. However, when excessive pumping occurred, the water levels in the northern part decreased during certain periods, reversing the water flow direction. This added to the complexity of the hydrodynamics of the site. The water budget indicated that canal irrigation was the main recharge flux at the basin scale and had a strong impact on groundwater quality. In such hard rock aquifers the saprolite plays a storage role due to its porosity, and the fissured zone provides the transmissive functions. When the water levels are shallow, the saprolite layer allows a regional groundwater flow.

Of all the wells monitored, W1 and W5 wells share similar hydrogeological conditions with the Musi River, where minimal variations in relation to water levels were observed. Thus, the canal plays only a passive role on the hydrogeology in the southern part of the watershed. The return flows induced mixing of the groundwater and canal water that was pumped for irrigation, especially in the middle potion of the watershed, which also contributed to a high groundwater recharge. Long term wastewater irrigation in the area has resulted in high salinity, compounded by agriculture run-off (e.g. nitrates, pesticides).

A number of pollution sources linked to agricultural activities (nitrates and pesticides in the northern part of the micro-watershed), water rock interactions (fluoride), and intensive agricultural practices with city wastewater were evident.

Given the number of contaminants found in groundwater, SAT has not contributed significantly to improving the groundwater quality, although the removal of some elements may have taken place. Further, while the geophysical studies performed showed zones with relatively deep weathering (potentially inducing high permeability), the limited size of the area, and continuous flooding due to agricultural activities may not allow sufficient retention time for effective treatment. Return flows may further enhance mineralization and facilitate fluoride release from the rocks.

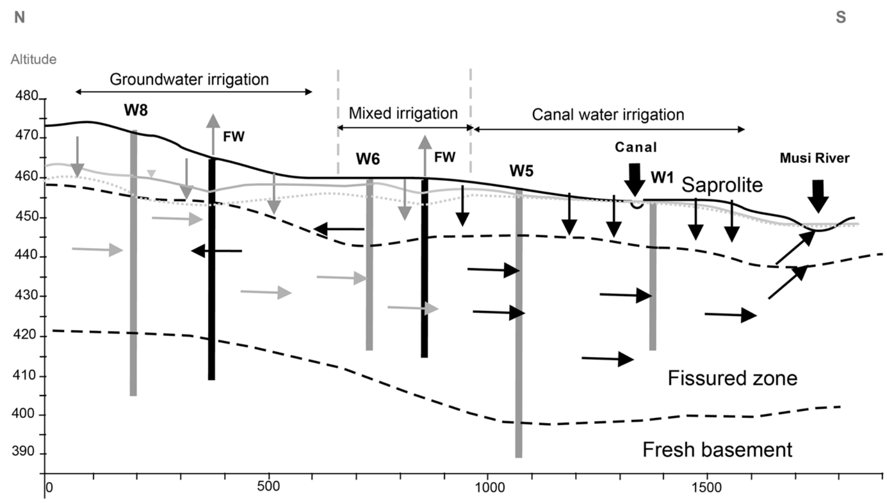

Figure 11.7 Conceptual model of groundwater flow and transport in the KSMWS. W8, W6, W5 and W1 are piezometric wells. The irrigation wells were used for agriculture. Dotted light grey line – pre-monsoon (June) water level; Light grey line – post-monsoon water level (November).

Overall, it is clear that in this natural setting, SAT may not be effective due to the flow patterns and short retention periods. Some preliminary tests for nitrates have shown that biogeochemical reactivity is dependent on hydrogeological and hydrogeochemical heterogeneity (Boisson *et al.*, 2013; McGuire *et al.*, 2002). Therefore, more studies under controlled conditions are required to understand the soil behaviour in remediation, and design small-scale engineered treatment systems that can be effective locally. Finally, although the potential for SAT may exist in the Musi River watershed, the complex hydrogeological conditions in this site were not conducive for the elimination of pollutants in a sustainable manner.

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