Groundwater Modeling for Sustainable Resource Management in the Musi Catchment, India

Sylvain Massuel[1], Biju George[2], Anju Gaur[1], Rajesh Nune[3]
[1] International Water Management Institute, Hyderabad, India.
[2] Department of Civil & Environmental Engineering, University of Melbourne, Australia-3010
[3] Jawaharlal Nehru Technological University, Hyderabad, India

Email: s.massuel@cgiar.org

Keywords: *Groundwater modelling, water resource, hard rock aquifer, semi-arid, India*

EXTENDED ABSTRACT

This study focuses on 11,000 km^2 of the Musi River sub-basin which is one of the main tributaries of the Krishna River, located in Andhra Pradesh (South India). The basin has a semi-arid climate with precipitation occurring during the rainy season, from June to October and with a total average of 710 mm yr^{-1}. During the past, the watershed development has led to significant changes in land use. Now, nearly 70% of the basin is cultivated of which 45% is irrigated. Around 60% of the water for irrigation is supplied by groundwater extraction. The number of bores in use has increased ten times from 1991 to 2001 and should currently exceeds 45,000 (on average, 1 active bore for 4.5 ha irrigated). The Musi Medium Irrigation project covers 12,500 ha and the Nagarjuna Sagar Left Canal supplies water for 43,000 ha downstream of the Musi sub-basin. Wastewater of mixed domestic and industrial origin is utilized to irrigate approximately >10,000 ha of paddy rice along the Musi River in peri-urban and rural Hyderabad. More than 1160 artificial percolation tanks had been built on the Musi catchment drainage network.

The preliminary analysis of more than 60 groundwater level time series widespread all over the sub-basin (from 1989 to 2004) shows a general long term depletion trend of the water table while no significant rainfall deficit was observed over the same period. The average rate of depletion is estimated as 0.18 m yr^{-1} with maxima in some areas of up to 0.40 m yr^{-1} (Figure 1). This situation can be explained mainly by groundwater exploitation, a consequence of the watershed development for agriculture. The Musi sub-basin is mainly covered by Archaean granites with Deccan Traps at the Eastern edge. As in a typical hard rock aquifer region, the yield of the bores decreases with depth due to the reduction of the fracture density. Hence the risk of water scarcity in case of a drought year is exacerbated. In order to assess aquifer renewable reserves and help groundwater management authorities, a fully distributed physical model of the aquifer has been calibrated and validated for a transient state experienced during 1989-2004 by using MODFLOW. The key variables such as aquifer storativity and transmissivity were determined by inverse fitting of simulated and observed groundwater levels.

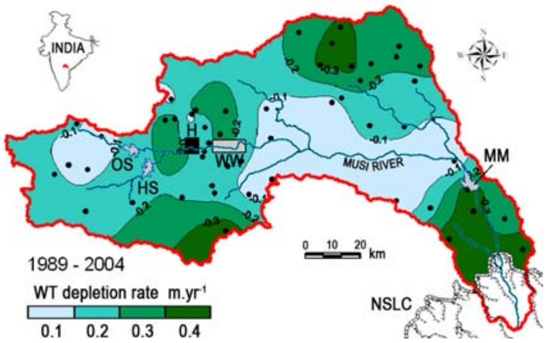

Figure 1. Water table depletion rate in the Musi sub-basin for the period 1989-2004. Observation wells are plotted with black points. [H] Hyderabad city; [WW] Wastewater irrigated area; [MM] Musi Medium irrigation project; [NSLC] Nagarjuna Sagar Left Canal; [OS] Osman Sagar Reservoir; [HS] Hymayat Sagar Reservoir.

Mean annual simulated recharge is 1176 Mm^3 (17% of total rainfall) while annual pumping is estimated at 1235 Mm^3. Simulated base flow is 23 Mm^3 while river leakage is less than 1 Mm^3. Among the total simulated annual recharge, groundwater irrigation return flow to the aquifer can be estimated at 370 Mm^3 (31%) and artificial recharge at 124 Mm^3 (11%). Natural recharge from rainfall accounts for 652 Mm^3 (55%). It is close to 9% of the total annual rainfall. The sustainable groundwater withdrawal yield over the period is around 1220 Mm^3 for the total basin. A deficit of 124 Mm^3 for the long term groundwater balance (16 years) justifies the observed depletion trend of the water-table of -0.18 m yr^{-1}.

1. INTRODUCTION

A countries historical economic development greatly impacts global water resources in terms of quality and quantity (Scanlon *et al.* 2007). In India, with the current rate of urban, agricultural and industrial expansion, and life style changes, water demands are growing and are likely to surpass supply from current sources. Under this overall condition of water scarcity, attempts at augmenting supply can create severe inter-sectoral water allocation conflicts. It is therefore crucial to improve our knowledge on all the potential water resources available. This will facilitate the development of appropriate strategies for a sustainable water resources management.

This paper is presenting the groundwater status of the Musi sub-basin in South-India as a case study for a preliminary groundwater resource modeling to predict the potential renewable storage that can be further used in water allocation models.

2. THE MUSI SUB-BASIN

The Musi sub-basin presents a wide spectrum of water uses, unique within the Krishna Basin. The basin has a semi-arid climate with precipitation occurring during the rainy season, from June to October and with a total average of 710 mm yr^{-1}. The Musi River is one of the main tributaries of the Krishna River and its catchment of 11,000 km^2 is entirely located in Andhra Pradesh, India (Figure 1). The Musi River originates in Anantha Hills, about 90 km to the west of the fast growing city of Hyderabad, and flows during the monsoon season from West to East across a flat topography (mean slope < 1%).

During the past four decades, watershed development in the Musi catchment has led to significant changes in land use. Now, nearly 70% of the basin is cultivated with 45% of the cropping area being irrigated. The Musi Medium Irrigation project covers 12,500 ha and The Nagarjuna Sagar Left Canal (NJSLC) supplies water for 43,000 ha downstream of the Musi sub-basin. Wastewater of mixed domestic and industrial origin is utilized to irrigate approximately >10,000 ha of paddy rice along the Musi River in peri-urban and rural Hyderabad. The upper catchment of the Musi River has been regulated by two dams, Osman Sagar and Himayat Sagar (Figure 1), which provide fresh water to Hyderabad city. In 2000, 1160 artificial percolation tanks built on the seasonal stream network were listed within the catchment. Further, around 60% of the water for irrigation is currently supplied by groundwater extraction. The number of bores in use has increased ten times from 1991 to 2001 and is estimated to currently exceed 45,000 while the number of dug wells in use is around 82,000.

The Musi sub-basin is mainly covered by Archaean granites with overlaying Deccan Traps at its Eastern edge. Groundwater resource is therefore mostly represented by typical unconfined shallow aquifers in hard rock which generally occupy the upper ≤20 m of the subsurface profile. These composite aquifers derive primarily from the geomorphologic processes of deep weathering and erosion. They can therefore be considered as a multi-layered system. Layers of a classical weathering profile of the region are detailed below from top to bottom (Marechal et al, 2004):

- unconsolidated weathered mantle (saprolite or regolith); from negligible to several tens of meters thickness. When saturated, this layer constitutes the reservoir of the aquifer.
- fractured-weathered layer; generally characterized by a fracture density that decreases with depth (Wyns *et al.*, 2004). This layer mainly assumes the transmissive function in the aquifer and is pumped by most of the bores
- fresh basement; which is permeable only locally where deep tectonic fractures are present.

3. METHOD

3.1. Data

Historical groundwater level data was collected from APGWD (State Ground Water Department). The data set includes pre- and post-monsoon levels for the period 1989-2004, drill-logs and interpreted pumping tests. A thorough data analysis of 97 observation wells and 47 piezometers led to the selection of 52 reliable time series for the reference piezometry determination. Regression models were fitted on each time series over the period 1989-2004 for the long term trend determination.

The aquifer seasonal and annual storage variations have been estimated for each year by the Water-Table Fluctuation method (WTF) well suited to the specific structure and hydrodynamic properties of hard-rock unconfined aquifers (Marechal *et al.*, 2006).

According to the drill-logs observations, the thickness of the weathered/laminated granites is on average 15 m and 20 m for the fractured zone. The layers are following more or less the topography which is explained by alteration processes (Wyns *et al.*, 2004).

3.2. Groundwater model setup

Groundwater flows are simulated with the finite-difference block centred groundwater model MODFLOW2000 (Harbaugh *et al.*, 2000). The active domain has been limited to the Aarchean granites, excluding basaltic aquifers of the Deccan Traps. The unconsolidated weathered mantle can be represented by a porous medium. Due to heterogeneity, discontinuity and anisotropy induced by the fracture networks, the hydrogeology of the fractured-weathered layer is more complex. For simplification purposes, it is assumed in this study that, at the Musi sub-basin scale, the fractured-weathered layer hydrodynamic behaviour can be equivalent to a porous medium.

The conceptual model consists therefore of two layers respectively for simulating weathered granites and fractured-weathered granites. Basement is characterised by the upper limit of fresh granites (Figure 2). Surface interpolation was determined through the interpretation of drill-logs and Digital Elevation Model (USGS EROS Data Center). After numerical tests, the grid resolution and time step calculations were fixed at 1 km^2 and 1 month (9692 cells), respectively.

Considering the general water-table flow direction, the groundwater basin limits coincide closely with the Musi catchment limits. Boundary conditions have been chosen with imposed nil flow on piezometric ridges and imposed calculated flow

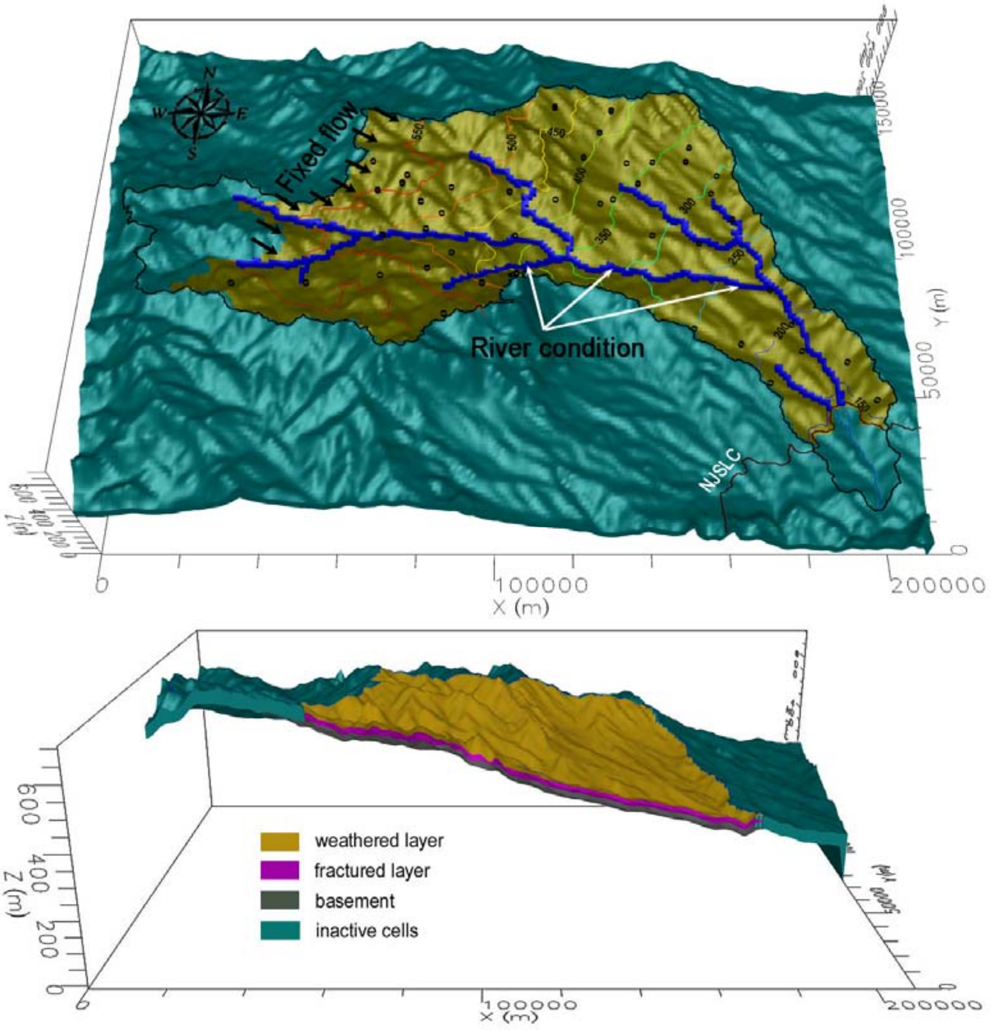

Figure 2. aquifer geometry and boundary condition of the groundwater model; 9692 active cells. Observation wells are plotted with black points (control points). NJSLC: Nargarjuna Sagar Left Canal.

(function of water-table gradient) for the connection with Deccan Traps (East). In the South, Nagarjuna Sagar Left Canal acts as a constant head limit and interaction with the Musi River is computed using Modflow River Package. The NJSLC command area of 430 km^2 is therefore excluded from the modelling domain (Figure 2).

Calibration was performed for the period 1995-2000 which is the period with the highest variation around the average in terms of water-table fluctuation. Validation was performed on periods 1989-1995 and 2000-2004. A total of 42 calibration points were selected for the representativeness and quality of their piezometric time-series. Calibration consists of inverse modelling of the groundwater levels. Best fitting between observed and simulated piezometry is obtained by minimizing value of the objective function (Root Mean Squared error) by tuning the three following parameters: total recharge (R), permeability (K) and storativity (S_y).

Calibrated variable R of the model can be interpreted as below:

$$R = GW_{rf} + SW_{rf} + R_{rain} + R_{art} \qquad (1)$$

where:
R is the calibrated total net groundwater recharge
GW_{rf} is the recharge from groundwater source irrigation return flow
SW_{rf} is the recharge from surface water source irrigation return flow
R_{rain} is the direct recharge from rainfall
R_{art} is the artificial recharge

The estimation of artificial recharge is based on the analysis of Landsat TM imagery of October 2000 (Biggs, unpublished). It has been found that 256 km^2 of the basin is covered by tanks ranging from 1 to 100 ha. Tank depths were taken from topographic maps. Accordingly, an average of 1 to 2 meters depth roughly yields a total 250 to 380 Mm^3 storage volume. Sukhija *et al.* (1997) and more recently Sharda *et al.* (2006) give percolation yields of artificial tanks ranging from 35% to 50%. Matching the grid between the 1160 identified tanks and the 1 km^2 grid of the model leads to 731 spatially distributed recharge cells, active during the whole year. To reproduce the increase in the number of tanks since 1989, the recharge rate follows the same exponential increase reaching a maximum after 2000.

Groundwater abstraction is simulated in the model by the MODFLOW Pumping Wells package. A total of 120,000 wells are in use over the modelling domain which means a density of around 12 wells per km^2. Actually, wells density is varying in space and time and ranges from 4 to 18 wells per km^2. Each cell of the grid model should therefore count more than one pumping well. For MODFLOW code, pumping wells are located in the centre of a cell, regardless of its actual location within the cell. For reducing time calculation, one "virtual" pumping well accounts for all the wells located in the same grid cell for simulating potential withdrawals. Pumping rate (P) has been assessed based on irrigation statistics, published census of the number of wells, land use from satellite images (Biggs *et al.*, 2006) and field survey. In 2005, 35 bores were gauged all over the Musi sub-basin and a discharge average of ~150 L min^{-1} has been found. Inquiries about more than 150 farmers' cropping practices indicate an average of 240 days of irrigation for paddy fields which is the dominant irrigated crop. Within the state, electricity is freely delivered to farmers for 5 to 7 hours a day. Relationship between groundwater withdrawal rate and cropping area determined by Dewandel *et al.* (2007) has been applied to the catchment usual land-use pattern. Paddy, sugar cane and other crops respectively roughly account for 77%, 5% and 8% of the irrigated area. Return flow from groundwater and surface irrigation to the aquifer (SW_{rf} and GW_{rf}) are also derived from the same study by calculating a general return flow coefficient weighted by the area of crops.

GW_{rf}, SW_{rf} and R_{art} variables are known as they are directly calculated using available data. Natural direct recharge from rainfall R_{rain} is function of annual total rainfall and is applied only during wet season (5 months in the model).

The groundwater direct evaporation (E) from the water table is evaluated according to a power law established for semi-arid areas that depends on groundwater level depth ($E = 71.9z^{-1.49} = 2.3$ mm).

4. RESULTS

4.1. Groundwater status

Depending on year, mean depth of the water-table varies for pre-monsoon between 11.3 m to 8.0 m below ground level and between 8.6 m to 5.0 m for post-monsoon. General flow runs from East to West following the topography (Figure 2) with piezometric gradients lower than 0.1%. The regression models fitted on the 52 piezometric time series show a long term depletion trend of the water-table. The calculated mean rate is -0.18 m.yr^{-1}, with high variability between 1 to 0.40 m.yr^{-1} depending of the observation well location. Origin of such variability can be attributed to (i) the aquifer hydrodynamic heterogeneity and/or (ii) the recharge intensity variations and/or (iii) the groundwater extraction wells per km^2. The depletion rate variability contour map is presented in Figure 1. High depletion rates occur in the North-East and downstream of the Musi Medium irrigation project where high groundwater abstraction rates can be expected due to supplemental irrigation. Hyderabad shows a lower depletion rate than its surroundings. Two assumptions are posibble: (i) the city could reduce the water-table depletion by the large water losses of the water delivery network (Van Rooijen *et al.*, 2005) or (ii) the city could increase the groundwater depletion rate on its outskirts due to high groundwater demand for domestic and industrial uses. Only a fine scale groundwater budget would be able to clarify this point.

Pre- and post-monsoon, groundwater levels rise on average by +2.89m per year, from 1.69 to 5.25 for 2004 and 1998 respectively. This value is the net groundwater balance for the monsoon period. It can be therefore considered as the groundwater renewable rate available for the following dry season. However, since aquifer storativity is unknown, fluctuations can not be converted directly into water volume values.

4.2. Groundwater modelling

Considering all the 42 control points, for the calibration period (1995-2000), the optimized objective function Root Mean Squared error (RMS) value is on average, 7.45 m (Figure 3). This result is quite acceptable regarding the range of piezometric data values (145 to 632 m). The good fit is confirmed by the low inter annual Normalized RMS value of 1.61%. Validation periods (1989-1994 and 2001-2004) show a slight increase of the RMS value (7.83 m) while Normalized RMS remains nearly the same (1.62%). The inter annual average of the absolute Residual Mean (average magnitude of the residuals noted hereafter “abs. RM”) is 6.23 m for the calibration period and 6.15 m for the validation period. Over the period 1989-2004 RMS is 7.68 m and abs. RM is 6.18 m (Figure 3). The model gives for the weathered granites layer: $K = 2.1\ 10^{-6}$ m.s^{-1} and $S_y = 16\%$, and for the fractured-weathered layer: $K = 8.6\ 10^{-6}$ m.s^{-1} and $S_y = 1.4\%$. These values are quite consistent with those evaluated from other studies on hard rock aquifers in the region (*e.g.* Marechal et al, 2004). The mean aquifer equivalent storativity (weighted by the volumes) is around 7% which means that the error in simulated water-table height should roughly not exceed 0.43 m.

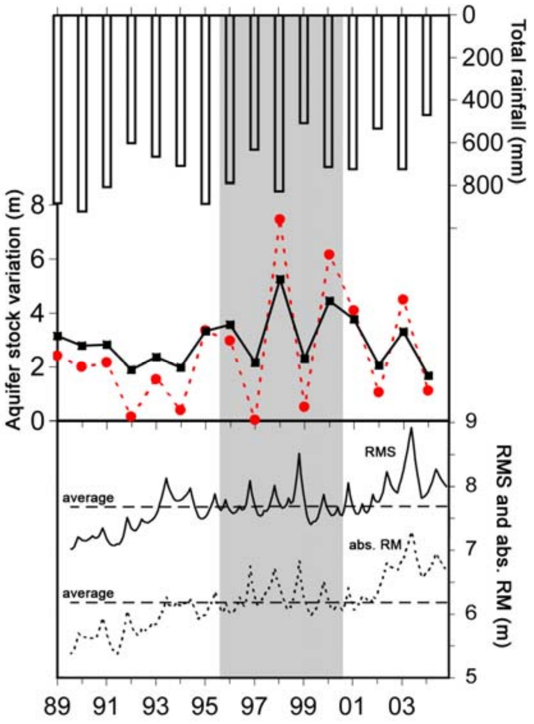

Figure 3. Aquifer annual stock variations for the monsoon season and total annual rainfall: comparison between WTF method (black line) and model simulation (red dashed line) with calibration period in grey overlay. RMS and absolute Residual Mean after model calibration; on average RMS = 7.68 m and abs. RM = 6.18 m for the period 1989-2004.

The mean inter annual total recharge is $R = 1176$ Mm^3 (17% total rainfall) while pumping is estimated at $P = 1235$ Mm^3 (Figure 4). Following equation (1), the different recharge components have been calculated and reported in Table 1. Among the 1176 Mm^3 of the simulated inter annual recharge, groundwater irrigation return flow and artificial recharge can be estimated at 370 Mm^3 (31% R) and 124 Mm^3 (11% R), respectively. Consequently, inter annual direct recharge from rainfall accounts for 9.4% of the total rainfall, *i.e.* 652 Mm^3 (55% R). The model calculates a lateral inflow to the aquifer of 4 Mm^3 and a lateral outflow of 33 Mm^3. Simulated base flow (B_f) is 23 Mm^3 while river leakage (L) is less than 1 Mm^3 (Figure 4). The very low leakage value may be due to the relative position of the water-table usually above the river bottom. Because of the non-perennial river flow, the major part of the surface runoff deep percolation occurs through artificial tanks and may also explain the low simulated leakage. Annual water-table evaporation is estimated at 16 Mm^3. Considering all groundwater inflows and outflows, total mean annual aquifer storage variation is -124 Mm^3. For a mean aquifer equivalent storativity of 7%, the simulated stock deficit explains the declining water-table trend of -0.18 $\mathrm{m.yr}^{-1}$. The sustainable groundwater withdrawal yield is therefore close to 1109 Mm^3 for the total basin.

	GW_{rf}	SW_{rf}	R_{rain}	R_{art}	R
Mm^3	370	30	652	124	1176
% rain	5.4	0.4	9.4	1.8	16.7
% *R*	31	3	55	11	100

Table 1. Total recharge components calculated values, inter annual average.

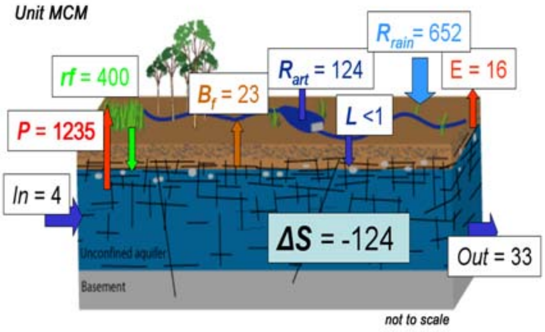

Figure 4. Simulated inter annual groundwater budget. [ΔS] stock variation; [L] river leakage; [P] pumping; [rf] total return flow from irrigation ($GW_{rf} + SW_{rf}$); [B_f] base flow; [R_{art}] artificial recharge; [R_{rain}] direct recharge from rainfall; [In] lateral inflow; [Out] lateral outflow; [E] evaporation.

5. DISCUSSION

For the Musi sub-basin hydrological context, WTF method provides reliable results for assessing relative groundwater availability for a given period. But in an area where rise in water-level is not rigorously only due to rainfall but influenced by nearby pumping well or return flow due to applied irrigation, effective recharge and aquifer storage can not be estimated accurately through this method. Even so, results can guide the modelling by comparing values qualitatively. This explains the bias that exists between the stock variations evidenced by WTF and the model (Figure 3), while the correlation between these two stock values is very good (R^2=0.924).

For simplification, the study did not take into account deep hardrock aquifer (tectonic fractures). Deep fracturation is not very frequent in the area but may account locally for a significant part of the groundwater resource provided that it can be tapped (spatially limited).

The main error on observed water levels originates from the error in the absolute elevation of wells (derived from GPS and DEM, *i.e.* 5 to 10 m). Consequently, in this paper, absolute water-table elevation in the sub-basin is indicative. However, this error is constant in time and specific to each well. When relative fluctuations of water-table are calculated, the error in well absolute elevation still exists, but is not involved as long as only the variation of elevation is considered. Therefore, dealing with stock variations less than 5 m is still relevant for this study.

According to equation (1), calibrating the R variable in the model is effectively, equivalent to calibrating recharge from rainfall (R_{rain}) as all other variables (GW_{rf}, SW_{rf} and R_{art}) are directly calculated using available data. For the validation period, because R_{rain} is frozen (9.4% of total rainfall), RMS is therefore highly dependant on the calculated values of GW_{rf}, SW_{rf} and R_{art}. Further, their contribution to R is increasingly important due to the expansion of the number of tanks and irrigated area. The best fit model is obtained for the period 1989-1993 whilst beyond the calibration period RMS tends to increase. This confirms the relevance of the calibrated R_{rain} value but also means that an important error arises from the knowledge of GW_{rf}, SW_{rf} and R_{art}. More statical tests such as multi-linear regressions would be needed to confirm this hypothesis.

6. CONCLUSION

The presented modelling is a preliminary result that gives an overview of the groundwater budget in the Musi sub-basin. Results are consistent enough to give better monthly groundwater resource estimates that other global water balance methods such as Water Table Fluctuation. It has been found that for the past 16 years, natural recharge from rainfall accounts for 9.4% of the total annual rainfall (652 Mm^3) while the sustainable annual groundwater withdrawal yield over the period is close to 1220 Mm^3 for the total basin. A deficit of around 124 Mm^3 for the long term groundwater balance justifies the observed depletion trend of the water-table of -0.18 m yr^{-1}.

Results presented by this paper illustrate that at the sub-basin scale, groundwater modelling in a hard rock semi-arid context can be a well suited tool for estimating general groundwater resource evolution. Linking with inter-sectoral allocation models for building future management scenarios can be therefore considered.

7. REFERENCES

Biggs T. W., M.K. Gumma, P.S. Thenkabail, C. Scott, G.R. Parthasaradhi and H Turral, (2006), Vegetation phenology and irrigated area mapping using MODIS time-series, ground surveys, and agricultural census data in Krishna river basin, India, *International Journal of Remote Sensing*, 27 (19), 4245-4266.

Dewandel B., J.M. Gandolfi, F.K. Zaidi, S. Ahmed and K. Subrahmanyam (2007), A decision support tool with variable agroclimatic scenarios for sustainable groundwater management in semi-arid hard-rock areas, *Current Science*, 92, 8, 1093-1102.

Harbaugh A.W., E.R. Banta, M.C. Hill and M.G. McDonald (2000), MODFLOW-2000, the U.S. Geological Survey modular ground-water model. User guide to modularization concepts and the Ground-Water Flow Process. *U.S. Geological Survey Open-File Report*, 00-92, 121.

Maréchal J. C., B. Dewandel, and K. Subrahmanyam, (2004), Use of hydraulic tests at different scales to characterize fracture network properties in the weathered–fractured layer of a hard rock aquifer, *Water Resour. Res.*, 40, W11508.

Maréchal J. C., B. Dewandel, S. Ahmed, L. Galeazzi and F. K. Zaidi, (2006), Combined estimation of specific yield and natural recharge in a semi-arid groundwater basin with irrigated agriculture, *J. Hydrol.*, 329, 281–293.

Scanlon, B. R., I. Jolly, M. Sophocleous, and L. Zhang (2007), Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality, Water Resour. Res., 43, W03437.

Sukhija B. S., D. V. Reddy, M. V. Nandakumar and Rama, (1997), A method for evaluation of artificial recharge through percolation tanks using environmental chloride, *Ground Water*, 35-1, 161-165.

Sharda V.N., R.S. Kurothe, D.R. Sena, V.C. Pande and S.P. Tiwari, (2006), Estimation of groundwater recharge from water storage structures in a semi-arid climate of India, *J. Hydrol.*, 329, 224-243.

Van Rooijen D.J., H. Turral, T.W. Biggs, (2005), Sponge city: Water balance of mega-city water use and wastewater use in Hyderabad, India, *Irrigation and drainage*, 54, S81-S91.

Wyns R., J. M. Baltassat, P. Lachassagne, A. Legchenko, J. Vairon, and F. Mathieu (2004), Application of SNMR soundings for groundwater reserves mapping in weathered basement rocks (Brittany, France), *Bull. Soc. Geol. Fr.*, 175(1), 21–34.