Improving assessment of groundwater-resource sustainability with deterministic modelling: a case study of the semi-arid Musi sub-basin, South India

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ABSTRACT

Since the 1990s, Indian farmers, supported by the government, have partially shifted from surface-water to groundwater irrigation in response to the uncertainty in surface-water availability. Water management authorities only slowly began to consider sustainable use of groundwater resources as a prime concern. Now, a reliable integration of groundwater resources for water-allocation planning is needed to prevent aquifer overexploitation. Within the 11,000-km² Musi River sub-basin (South India), human interventions have dramatically impacted the hard-rock aquifers, with a water-table drop of 0.18 m/yr over the period 1989-2004. A fully distributed numerical groundwater model was successfully implemented at catchment scale. The model allowed two distinct conceptualizations of groundwater availability to be quantified: one that was linked to easily quantified fluxes, and one that was

34 more expressive of long-term sustainability by taking account of all sources and sinks.
35 Simulations showed that the latter implied 13% less available groundwater for exploitation
36 than did the former. In turn, this has major implications for the existing water-allocation
37 modelling framework used to guide decision makers and water-resources managers
38 worldwide.
39

41 **1 Introduction**

42 A country's historical economic development greatly impacts its water resources in terms of
43 quality and quantity (Scanlon et al. 2007). In India, with the current rate of urban, agricultural
44 and industrial expansion, and life style changes, water demands are growing and are likely to
45 surpass the supply from current sources. Since the 1850s, the Krishna basin (Figure 1) – and
46 India as a whole – has witnessed a remarkable development of irrigation and water use with
47 the construction of large dams in the main rivers, minor irrigation schemes in secondary
48 upstream catchments and a booming groundwater economy (Shah, 2009; Venot, 2009).

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50 With increasing interconnectedness of users due to basin closure – most or all available water
51 is committed and river discharge falls short of meeting environmental functions (Molle et al.,
52 2007) – upstream catchment development is one of the main causes of declining downstream
53 water availability (Batchelor et al., 2003). The minor irrigation census of 2001 estimated that
54 about 175,000 minor irrigation structures (typically tanks and check dams) were in use in the
55 Krishna basin (Venot, 2009). This leads to lower, unreliable and late canal water deliveries in
56 large irrigation schemes downstream, pushing farmers to increasingly resort to groundwater
57 and increases the likeliness of local and basin conflicts (see Venot et al., 2010; Shah, 2009).
58 Firstly used as an alternative source to scarce surface water, groundwater has now become the
59 main water resource for irrigation (e.g. in the Musi sub-basin Figure 2), because of its
60 availability and reliability when compared to surface water.

61

62 Despite changes in the relative and absolute importance of different water sources, water
63 allocation procedures in India have always dealt with surface water only. The Krishna Water

64 Disputes Tribunal set up by the Government of India in 1969 to share the Krishna waters
65 between the three riparian states of Andhra Pradesh, Karnataka and Maharashtra for example
66 stated that the three states “will be free to make use of underground water within their
67 respective territories in the Krishna river basin [and that] use of underground water shall not
68 be reckoned as use of the water of the river Krishna” (GoI-KWDT, 1976). The recent Cauvery
69 award apportioning the waters of the Cauvery River makes a similar statement (GoI-CWDT,
70 2007).

71

72 Groundwater is available in quantity at low cost (subsidized electricity) and is individually
73 controlled by farmers. A direct consequence is the overexploitation of the shallow hard-rock
74 aquifer in the sense defined by Custodio (2002): “the total amount of abstraction from an
75 aquifer is, or will be close to, or greater than the total recharge over several years”. In the
76 Krishna basin, groundwater abstraction and depletion led to streamflow reduction (Biggs et
77 al., 2007). This highlights the urgent need for accounting for groundwater resources to
78 achieve sustainable water allocation planning.

79

80 Introducing groundwater resources in water allocation plans is complex and requires a holistic
81 and integrated approach to obtain a sustainable, efficient and equitable outcome (Rogers &
82 Fiering, 1986; Jakeman and Letcher, 2003; Letcher et al., 2004, 2007; Jacobs and Holway,
83 2004; Babel et al., 2005; Cutlac et al., 2006). Sustained use of groundwater resources entails a
84 long-term evaluation of the hydrologic implications of given development strategies.
85 Decisions made on withdrawals may ultimately affect surface water resources, but many years
86 can be necessary for these effects to be realized. The hydrological challenge for addressing
87 issues of sustainability is therefore to assess the long-term impact of groundwater extraction.

88

Groundwater resources can be assessed using numerical models or water balance methods, but a trade-off has to be found between data availability, method complexity and expected relevance of outputs. Alternatives can be developed for specific cases (e.g. in Australia, Bekesi et al., 2009), where adaptive short to medium-term planning strategies consist in correcting allocations based on groundwater storage depletion. Determining the appropriate groundwater safe or sustainable yield (e.g. Domenico, 1972; Alley and Leake, 2004; Devlin and Sophocleous, 2005; Kalf and Woodley, 2005; Sophocleous, 2005) remains challenging for any of the methods but almost always the best way to quantify the elements needed is by means of a numerical groundwater flow model.

In India, the potential development of a groundwater basin is categorized based on the notion of renewable recharge, i.e. amount of water that seeps through soil into the aquifer. The objective of policy makers is to limit groundwater use to the rate of renewable recharge thus considered as the groundwater sustainable yield. Within complex hydrological systems, this choice is driven/justified by data scarcity and the need for a simple standardized determination protocol. This explains the common use of methods such as global water accounting, water-table fluctuation or rainfall-recharge relationship, which are easier to implement than numerical modelling but where complex non linear interactions of the water balance components are unspecified (e.g. Sethi et al, 2002, 2006; Naik and Awasthi, 2003; Thomas et al., 2009). Numerical modelling is data consuming and requires good knowledge of hydrological processes. But in some favourable cases, where those requirements are met, it may be worthwhile doing in order to produce a more consistent evaluation of the groundwater sustainable yield.

The present study is focusing on the Musi River sub-basin in South India. It aims to improve the relevance of sustainable groundwater-resource estimates, using a numerical modelling method spanning over 15 years of observation. Accordingly, the name “recharge yield” was given to the standard by which pumping does not exceed renewable recharge, and the name “discharge yield” was given to the standard by which pumping does not diminish natural discharge (to surface water or to the boundaries of the basin under study), indexed to a given time period. After characterization of the hard-rock aquifer system, a numerical groundwater model was developed at the basin scale (using the numerical code MODFLOW-2000). The groundwater long-term yield, under stresses from irrigation pumping, was simulated by the model. Two water allocation scenarios were run, both with the simulated discharge yield and the estimated recharge yield. The implications of considering these different evaluation methods of groundwater availability in water management plans were discussed.

2 The Musi sub-basin

2.1 Hydrologic and hydrogeologic settings

The Musi River is one of the main tributaries of the Krishna River and its catchment (about 11,000 km²) is entirely located in the state of Andhra Pradesh (Figure 1). It originates in the Anantha Hills, about 90 km to the west of the fast growing city of Hyderabad, and flows intermittently during the monsoon season from West to East across a flat topography (mean slope < 1%). The climate is semi-arid and the monsoon season annually provides on average 710 mm of rainfall in only 5 months, from June to October. As a result of this typical distribution of precipitation, groundwater recharge occurs only during the wet season.

The Musi sub-basin presents a wide spectrum of water uses: irrigated agriculture is the major use but urban and industrial demands around Hyderabad city are fast growing and have triggered large scale water transfers from the neighbouring Godavari basin and from large scale irrigated area downstream, in the Krishna basin (Celio et al., 2010). Over the last four decades, the Musi catchment has witnessed dramatic changes in land use. Nearly 70% of the basin area is currently cultivated with 45% of the cropping area being irrigated.

The upper catchment of the Musi River has been regulated by two dams since the 1920s, Osman Sagar and Himayath Sagar (Figure 1), which provide fresh water to Hyderabad city. The Musi Medium Irrigation project covers 20,000 ha and the Nagarjuna Sagar Left Canal (NJSCL) supplies water for 43,000 ha in the downstream reaches of the Musi sub-basin (Figure 1). Wastewater of mixed domestic and industrial origin is utilized to irrigate approximately 10,000 ha of paddy rice, vegetables and fodder crops along the Musi River downstream of Hyderabad. In 2000, 1160 artificial reservoirs built on the seasonal stream network were inventoried within the catchment (Biggs, IWMI, unpublished report, 2005; Figure 1).

The net area irrigated by groundwater supply increased by 35% between 1989 and 2004. During this period, the net area irrigated by surface water supply reached a maximum and decreased (Figure 2). In 2004, around 75% of the water used for irrigation was supplied by groundwater, illustrating the shift from surface to groundwater source. The number of pumping wells in use increased ten-fold from 1991 to 2001 and is estimated to currently exceed 45,000 while the number of dug wells in use is around 82,000 (Handbook of Statistics, 1992 to 2008).

The Musi sub-basin is characterized by Archaean granitic rocks with overlying Deccan Traps at its extreme Western edge. Groundwater resources are therefore mostly represented by typical unconfined shallow aquifers in hard-rock in the upper 20-40 m of the subsurface profile (Dewandel et al., 2006). The general flow runs from West to East (Figure 1) following the topography with piezometric gradients lower than 0.1%.

These composite aquifers derive primarily from geomorphologic processes of deep weathering and erosion of fresh granites (Wyns et al., 2004; Lachassagne et al., 2011). The weathering expands from the surface down to the ground and creates distinctive levels of alteration parallel to the topography. In this region, according to observations in drill-logs and dug wells, the uppermost level consists of unconsolidated laminated-weathered mantle (saprolite or regolith) generally 10-15 m thick. When saturated, this layer constitutes the storage capacity of the aquifer, with a network of mainly subhorizontal fissures partially filled up by clayey minerals. The underlying level consists of fractured weathered granite, 15-20 m thick, that assumes the transmissive function of the aquifer; most of the wells are pumping from this layer. It is characterized by a density of hydraulically conductive fractures that decreases with depth. Underlying is the unfissured fresh granite (bedrock) which can be considered as totally impermeable at the watershed scale due to the absence of major tectonic structure (Figure 3, Maréchal et al. 2004; Dewandel et al, 2006).

2.2 Groundwater status

Historical groundwater level data were collected from the Andhra Pradesh Ground Water Department (APGWD). The data set included pre- and post-monsoon levels from May 1989 to November 2004, drill-logs and interpreted pumping tests. A thorough analysis of data from 97 observation wells and 47 piezometers led to the selection of 52 reliable time series for the

reference piezometry determination. Linear regression models were fitted on each time series over the period 1989-2004 for assessing long term trends (Massuel et al., 2007). The aquifer seasonal and annual storage variations have been estimated for each year by a water-table fluctuation method that is well suited to the specific structure and hydrodynamic properties of hard-rock unconfined aquifers (Maréchal et al., 2006).

The general trend of the water table levels over the period 1989-2004 is presented in Figure 1 and Figure 3. For the 52 observation points, the annual mean depth to the water table ranges from 11.3 m to 8.0 m below ground level in pre-monsoon season and from 8.6 m to 5.0 m in post-monsoon. The inter-annual mean fluctuation between the 2 seasons is 2.9 m, varying from 1.7 to 5.3 m for 2004 and 1998 respectively. The water table shows a long term declining trend of 0.18 m/yr on average, with high spatial variability ranging from 0.1 to 0.4 m/yr (Figure 1). The origin of such variability over a 15-year period can be attributed to (i) the aquifer hydrodynamic heterogeneity and/or (ii) the groundwater abstraction intensity. High rates of water-table decline occur in the North-East and downstream of the Musi Medium irrigation project where groundwater irrigation is also important. The rate of decline in Hyderabad city is lower than in its surroundings. The possible explanations are that (i) water losses in the city water delivery network could reduce groundwater resource depletion (Van Rooijen et al., 2005) and/or (ii) the groundwater depletion rate on the city's outskirt is the result of high groundwater demand for domestic and industrial uses.

3 Groundwater model settings

3.1 Conceptual model

Groundwater flow in the Archaean granitic aquifer was simulated with the finite-difference block-centred groundwater model MODFLOW-2000 (Harbaugh et al., 2000). Two numerical layers parallel to the ground surface were created according to the weathering profile (Figure 4). The unconsolidated laminated/weathered mantle was represented as a porous medium (15 m thick). According to Wyns et al. (2004), the discontinuity and anisotropy induced by the fracture networks in the fractured/weathered layer are minimized at large scale. At smaller scale (80 km²) Lubczynski and Gurwin (2005) showed that a transient numerical model within fractured granites could be successfully implemented with MODFLOW. At the Musi sub-basin scale the fractured/weathered layer was therefore also represented as a porous medium (20 m thick). The basement was characterised by the upper limit of the fresh granites (Figure 3). The surfaces of the layers were determined based on the digital elevation model of the basin (3 arc-seconds SRTM, USGS EROS Data Center) and by interpreting and correlating 20 good quality drill-logs evenly distributed on the modelling domain. Dewandel et al. (2006) showed with 50 drill-logs on a 30-km² basin that using topography information with weathering thicknesses to delineate the contour of layer surfaces was a good approximation.

According to observed hydraulic properties of the aquifer and the smooth hydraulic gradient, a trade-off between precision and complexity was obtained with a grid resolution of 1 km² (9692 cells) and monthly time step calculations.

3.2 Boundaries

Considering the general water-table flow direction, the northern and southern limits of the Musi River surface catchment coincide closely with the groundwater basin, with flow lines aligned at right angles to the equipotential lines (Figure 4). This matching is typical in hard-rock regions and at such scale (Marechal et al., 2006). Boundary conditions were set there as no-flow. On the western side, the eastern extent of the overlying Deccan Trap lavas was taken as the limit of the modelling domain. This limit was set as a specified flow boundary, calculated for each time step as a function of the piezometric gradient and saturated thickness. The South-East boundary was defined by the water level in the Nagarjuna Sagar Left Canal (NJS LC) as a constant head limit.

3.3 Abstraction

Groundwater abstraction was considered in the model using the MODFLOW Pumping Wells package. Pumping well characteristics (coordinates, screen, discharge...) were based on irrigation statistics (AP Handbook of Statistics, 1989-2004), published census of the number of wells, land use from satellite images (Biggs et al., 2006) and field inquiries. The evolution of the total number of wells in use (dug wells + pumping wells) over the modelling domain started from 50,000 in 1989 and increased exponentially to reach around 127,000 in 2001. The density was varying in space and time from 4 to 18 wells per km².

Within the state, electricity is freely delivered to farmers for 5 to 7 hours a day and the actual pumping time was found to be close to 6 hours a day on average over the year (Massuel et al., 2008). From 2005 to 2008, 50 bores were gauged all over the Musi sub-basin and an average discharge of 9 m³/h was observed for around 240 days of irrigation per year. To cross validate those observations, surveys with nearly 150 farmers were carried out to assess their cropping

practices. Paddy, sugar cane and other crops were respectively accounting for 77%, 5% and 8% of the irrigated area. A good correlation between area irrigated under paddy crop and pumping rates was obtained (correlation coefficient $r = 0.82$), matching the relationship between groundwater withdrawals and cropping area determined by Dewandel et al. (2008). Pumping rates (P) were then distributed on the model grid in accordance with the catchment land-use pattern.

The groundwater direct evaporation (E) from the water table was evaluated according to a power law established for semi-arid areas that depends on groundwater level depth ($E = 71.9z^{-1.49}$; Coudrain-Ribstein et al., 1998). This relationship shows that the water table is too deep (>8 m) to allow any significant direct extraction from the root zone at the basin scale (same result obtained by Marechal et al. 2006).

3.4 Recharge

The recharge parameter in the groundwater model was defined as the vertical recharge that was effectively reaching the water table (i.e. after the interception by the root zone), that is, the net recharge (R_{net}), and was expressed as:

$$R_{net} = GW_{rf} + SW_{rf} + R_{rain} + R_{art} \quad (1)$$

with

GW_{rf} the excess irrigation return flow from groundwater source;

SW_{rf} the excess irrigation return flow from surface water source;

R_{rain} the direct recharge from rainfall;

R_{art} the artificial recharge from tanks.

The method based on crop-wise irrigation return flow coefficients developed by Dewandel et al. (2008) was used for estimating the excess irrigation seepage from surface and groundwater sources (SW_{rf} and GW_{rf} , respectively). The annual cropping pattern was inferred from land use data available in handbooks of statistics and correlated with the normalized difference vegetation index (NDVI) classification from Landsat TM imagery.

The estimate of artificial recharge R_{art} was based on an analysis of Landsat TM imagery of October 2000 (Biggs, unpublished, details as previous). It was found that 256 km² of the basin was covered by 1160 tanks ranging from 1 to 100 ha. An average of 1 to 2 meters depth was determined by field investigations of the reservoirs. 225 new tanks were constructed between year 2000 and 2003. This estimate was close to the reported numbers of existing tanks in the yearly district-level minor irrigation statistics which were used for the rest of the modelling period (i.e. 1989-1999). Sukhija et al. (1997) and more recently Sharda et al. (2006) and Massuel et al. (Massuel S, Perrin J, Mascre C, Wajid M, IWMI, IFCGR, unpublished paper, 2008) reported that percolation yields of artificial tanks range from 35% to 61% of the total inflow. Groundwater flow to small reservoirs has never been reported (because of the depth of the water table) and was therefore neglected.

Relationships between the total annual rainfall (P_{tot}) and the fraction that directly percolates to the aquifer (R_{rain}) have been established in the region using various methods (e.g. Rangarajan & Athavale, 2000). However, R_{rain} was not evaluated using one of the published relationships. Instead, it was decided to calibrate the net recharge R_{net} (see section *Calibration, validation and sensitivity*) from which R_{rain} was then inferred, GW_{rf} , SW_{rf} , and R_{art} being specified (Equation 1). The strategy was to cross validate the groundwater model calibration. The affine

transformation linking P_{tot} to the inferred R_{rain} was adjusted by linear regression and checked with estimates of other authors for consistency (see section 4.1).

3.5 Interaction with the Musi River

Owing to the semi-arid climate and the numerous tanks erected across the drainage system, the Musi River flow is intermittent. This means that connections with the aquifer are sporadic in space and time. Resulting flow transfers were therefore expected to be of the second order in the groundwater budget compared to percolation through reservoirs as estimated in section 3.4. Exchange between groundwater and surface-water was simulated as a 3rd-order head-dependent boundary using MODFLOW River Package (RIV). The model was populated based on three gauging stations located at each major reservoir in the river basin and at a fourth station located a few kilometres upstream of the confluence between the Musi and the Krishna rivers (George et al., 2007).

3.6 Calibration, validation and sensitivity

The period 1995-2000 was chosen for calibration because it encompassed the largest variability of water table fluctuation amplitudes. Calibration consisted of the inverse problem solving of the groundwater levels. Best fit between observed and simulated piezometry was sought by manually tuning the parameters of the model in order to minimize the root mean squared error function (RMSE). The 40 wells, out of the selected 52, that belong to the modelling domain were used as observation points.

Calibrating the model implied tuning the net recharge R_{net} and hydraulic properties of each layers of the model, namely the permeability (K) and the specific yield (S_y). Because, at the basin scale, granites were assumed to be homogeneous (see section 3.1), a single value for characterizing each layer of the model was considered. Values were constrained by the ranges

of hydraulic properties of the Archaean granite measured in the past few years in the basin (e.g. Maréchal et al., 2004, 2006; Dewandel et al., 2006; Chandra et al., 2008). Mean values of the measured hydraulic conductivity of the laminated and fractured layers were found to be 4×10^{-6} m/s (from 7×10^{-7} to 3×10^{-5} m/s) and 2×10^{-5} m/s (from 5×10^{-6} to 9×10^{-5} m/s), respectively. Measured total porosity values ranged from a minimum of 0.001 in the fractured layer to a maximum of 0.20 in the laminated weathered layer.

A steady state model was run for the year 1995 (best dataset) on which the transient model was implemented over the period 1990-2004 and validated over the periods 1990-1994 and 2001-2004. The water table of May 1989 was used as initial level and the year was not accounted in the budget. The model was run using the calibrated parameters, and the simulated hydraulic heads were compared to the 40 observation points with the RMSE function. Sensitivity tests were conducted by changing successively one of the three parameters around its calibrated value, keeping other parameters constant. Results give the sensitivity of the model outputs to the selected parameter.

4 Results and Discussion

4.1 Model calibration, validation and sensitivity

The calibration process resulted in an acceptable compromise between constraints on range of parameters and accuracy. The optimized mean RMSE value for the 40 control points was 4.8 m over the calibration period (1995-2000) whereas the mean absolute magnitude of the residuals (noted hereafter “abs.RM”) was 4.5 m (geometric means). The spatial distribution of residuals was fairly uniform. The mean normalized RMSE value (i.e. the RMSE divided by the maximum difference in the observed head values) of 1.36% confirmed the acceptable fit

between the observed and simulated heads as displayed on Figure 5 for the 40 observation wells.

The corresponding calibrated parameter sets were $K = 8 \times 10^{-6}$ m/s and $S_y = 0.17$ for the laminated/weathered granites layer and $K = 2 \times 10^{-5}$ m/s and $S_y = 0.0015$ for the fractured/weathered layer. These values are in the range of those evaluated from other studies on hard-rock aquifers in the region (see section 3.6).

The consistency of the simulated groundwater levels remained stable over the validation period (see Figure 5). The mean RMSE increased only by 0.4 m and the mean abs.RM by 0.2 m for a mean normalized RMSE of 1.37%. Figure 6 illustrates the change of RMSE and abs.RM over the full period of simulation along with annual water budget main outputs. Both indicators (RMSE and abs.RM) increased until 1993 and then stayed fairly stable over the remaining period.

The calibrated mean interannual net recharge was 1350 Mm^3 (139 mm), showing high variability from 900 to 1900 Mm^3 (Figure 6, calibration period). Based upon Equation (1), R_{rain} was calculated and the linear relation with the total annual rainfall (P_{tot}) was adjusted giving $R_{rain} = 0.192 P_{tot} - 61$ ($r = 0.996$). As shown in Figure 7, this relation is consistent with existing results obtained from field experiments using a tritium tracer technique in the region (e.g. Rangarajan & Athavale, 2000) and illustrates the consistency of the calibrated net recharge R_{net} .

Sensitivity tests confirmed the relative stability of the model outputs regarding calibrated parameter ranges. For instance, a relative variation of +10% of the permeability of both layers

generated an increase of the inflow of 3% and a drop of the outflow of 1%. The maximum variation induced on annual storage remains below 6% and below 1% for the simulated exchange with the river. The absolute mean variation of the simulated water table was 0.04 m. Modifications of the specific yield by $\pm 10\%$ of the absolute value were not generating any change in any of the water budget terms more than 1.5%. The absolute mean variation of the piezometric heads remains below 0.05 m during the tests.

The absolute magnitude of the residuals may appear at first glance excessive compared to the change in groundwater storage. Actually, 40% of observed groundwater levels are within or close to the fractured/weathered zone in the pre-monsoon period and that is also where most of the error lies (77% of the absolute total error). Indeed, owing to the very low capacitive property of the fractured/weathered layer ($S_y=0.0015$), any variation of groundwater levels involves very little flow into or out of storage (e.g. 1-m variation of level involves 1.5 mm of water).

The relative reliability of the model is a good argument for demonstrating that distributed numerical modelling can be implemented in hard-rock aquifers at catchment scale, as pointed out by Lachassagne et al. (2001).

4.2 Water budget

The magnitudes of the simulated water budget terms are very variable with a very low baseflow component compared to more humid basins. This reflects typical conditions in semi-arid basins that are heavily pumped for irrigation. Over the full period, the simulated mean interannual net recharge was $R_{net} = 1170 \text{ Mm}^3$ (equivalent to net recharge rate of 121 mm/yr) (Table 1; see Figure 6 for annual values and Figure 8 for spatial distribution). Following Equation 1, the different recharge components have been calculated and reported in Table 2.

Mean interannual excess irrigation return flow and artificial recharge were estimated at 400 Mm³ (41 mm) and 122 Mm³ (13 mm), respectively. Thus the mean interannual direct recharge from rainfall (R_{rain}) was estimated at 648 Mm³ (67 mm, 9.4% of P_{tot}), which is consistent with other studies (e.g. Perrin et al. 2012).

The mean interannual groundwater pumping was estimated at $P = 1235 \text{ Mm}^3$ (127 mm). Abstraction by pumping in the model started to slowly decrease after 1998 (Figure 6). Indeed, owing to lowered groundwater levels, the number of dried out wells increased and lowered the abstraction capacity despite the increase in number of wells. However, this phenomenon was probably exaggerated in the model since the net irrigated surfaces by groundwater source continued to grow until 2000. Actually, when observed in the field, discharge from shallowest (and usually oldest) pumping wells becomes intermittent with lowered groundwater levels (due to excessive drawdown the pump sucks air through swirls) whilst they are totally non productive in the model. In addition, the growth of the number of wells in use is reduced after 1998. In 1987, the density of wells in use in the Musi basin climbed from 3.4 well/km² to 8.5 well/km² in 1998 (i.e. +0.50 well/km²/yr), while it was 9.3 well/km² in 2001 (i.e. +0.25 well/km²/yr).

The model simulated a marginal mean interannual groundwater lateral inflow to the aquifer of 4 Mm³ (<1 mm) that can be supplied by the overlying Deccan Traps. A larger mean interannual lateral outflow was calculated (33 Mm³; 3 mm), the aquifer being drained by the NJS LC. However, the simulated annual lateral outflow was dominant in 1990 (170 Mm³) and decreased drastically as a result of reduced head gradients with the NJS LC (owing to the groundwater level drop).

Owing to the depth of the water table, the change in groundwater evaporation was insignificant. Its contribution to the water budget was estimated at -16 Mm^3 (-2 mm), on average.

The mean simulated groundwater interannual contribution to the river baseflow was 23 Mm^3 (2 mm) whilst the river leakage was less than 1 Mm^3 . It was difficult to clearly estimate the validity of those results, but the values were consistent with discharges measured at the 4 gauging stations (10% of the averaged interannual runoff, period 1989-2001). The dominant flow movement was therefore from the aquifer to the river, which was confirmed by the piezometric contour map illustrating the drainage of the aquifer by the river, typical in hard-rock regions (Figure 1).

Finally, over the period 1990-2004, the mean interannual change in aquifer storage was negative and estimated at $-132 \text{ Mm}^3/\text{yr}$ ($14 \text{ mm}/\text{yr}$), in accordance with the observed overall long term water-table decline of $0.18 \text{ m}/\text{yr}$.

4.3 Determining the sustainable yield

The intention here was not to re-examine the definition of groundwater sustainability, but to show, by a case study, how numerical groundwater modelling can increase the significance of the hydrological component of groundwater-sustainability assessment and thus modify its impact on water resource management planning.

In the Musi sub-basin, the groundwater sustainable yield is usually defined based on simple water accounting. The abstraction is meant to be sustainable when it is less or equal to the renewable component of the groundwater resource, i.e. the net recharge (R_{net}). This concept

may be satisfying to classify groundwater basins according to their level of abstraction at one point, but it is too limited to define long-term allocation strategies and future water management plans.

Distributed numerical modelling is meant to improve the meaning of sustainable yield as it considers a more holistic picture of the long-term groundwater sustainability than the simple renewable recharge. The model explicitly simulates all sources and sinks and therefore allows estimation of the effect of pumping on natural discharge rates to surface water or across basin boundaries and what pumping rates are possible in order to sustain the other discharge rates at a specific level. The groundwater-resource sustainability can be assessed and quantified as the amount of water that can be withdrawn from the aquifer system (for irrigation purposes in the present case study) without inducing any long-term depletion of the groundwater storage. This so-called discharge yield is constantly adjusted considering the impact of extractions and uses on the reserves through the different water transfer processes (irrigation seepage, groundwater baseflow etc.). This is possible because the model integrates time and space variability of the system to reproduce appropriate redistribution of groundwater regarding, for instance, river distance, water-table drop rate or pumping rates.

The discharge yield over the full period was determined in the model by tuning pumping rates and return flow from excess groundwater irrigation (GW_{rf}), every other thing being equal, in order to obtain a mean interannual change in the aquifer storage close to zero. Simulations showed that a mean interannual pumping rate of 1017 Mm³/yr did not affect the aquifer storage over the 15-year period and sustained the water table at its initial level (1990).

The annual net recharge is reported in Figure 9 (recharge yield) along with the simulated discharge yield for comparison with actual pumping. It appears that up to 1998, mean

interannual actual pumping exceeded the simulated mean interannual discharge yield by 40%. This period of overdraft was responsible for 60% of the total water table drop over the full period. Over the 15-year period, mean interannual actual pumping exceeded the simulated mean interannual discharge yield by 20% and stabilised close to the mean interannual discharge yield only from 2002 onward.

To cover the estimated long term deficit in groundwater storage of 132 Mm³/yr and sustain water table levels (and thus protect natural discharge), the actual mean interannual groundwater pumping (1235 Mm³/yr) would need to be cut by 18% according to the simulated discharge yield (1017 Mm³/yr), and by only 5% according to the estimated recharge yield (1170 Mm³/yr). With respect to the non-linearity of multi-scale groundwater flow processes, reduction in unit volume of pumped groundwater implies different changes in storage depending on time, space and amplitude conditions of the reduction. The distributed numerical model showed that integrating those conditions can have significant impact on evaluation of groundwater availability (e.g. 13% discrepancy in the present case study). The next section investigates the implications of these discrepancies for designing water allocation procedures.

5 Implications for water allocation

Most simulation-based water allocation models operate on the principle of water balance. The commonly used models in this category are MODSIM-DSS (Fredericks et al. 1998), MIKE-SHE (DHI, 2008), WEAP (Yates et al. 2005) and REALM (Perera et al. 2005). The Resource Allocation Model (REALM) was selected because of its adaptability, making it adequate for the current study. REALM is a generic water allocation tool for long-term operational

497 planning. It can be used for both urban and agricultural water supply systems to analyse
498 alternate water allocation policies (Perera et al. 2005; George et al. 2011a).

499
500 REALM allocates water resources based on supply and demand scenarios and is based on the
501 principles of water balance. REALM considers user defined penalties to decide the
502 preferential supply paths which are used in a network linear optimization algorithm to solve
503 the water allocation problem (Perera et al. 2005; George et al. 2011a). The system
504 components are represented as a network of nodes (such as reservoirs, groundwater basins,
505 townships, agriculture, industries, confluence etc.) and links connecting the nodes (rivers,
506 pipelines, canals etc.). The water demand component is estimated externally and input into the
507 model. Once water demand and supply has been established, the allocation tool is used to
508 match demand with available supply in the catchment.

509
510 For the Musi sub-basin, industrial, agricultural, domestic, power generation and
511 environmental demands are considered. Detailed water allocation modelling was carried out
512 in the Musi catchment and is described in George et al. (2008, 2011a). In the model,
513 groundwater was accounted for as a simple “reservoir” and the amount of water that can be
514 supplied by the reservoir – groundwater availability – was determined by the “inflow”
515 parameter (

516 Figure 10). Thus, the groundwater availability was estimated in two different ways: first by
517 considering the recharge yield (“recharge yield” scenario) and second by considering the
518 discharge yield (“discharge yield” scenario). The groundwater reservoir was then connected
519 to different demand centres (e.g. groundwater irrigated areas). Groundwater demand is
520 approximated to actual pumping.

The monthly volume of water supplied and shortfall generated by the REALM model were used to calculate the annual values for each demand node for the simulated time series. The simulations produced a mean interannual unmet demand of 150 Mm³/yr for the “discharge yield” scenario and 105 Mm³/yr for the “recharge yield” scenario. The water deficit occurred mainly in the dry season (up to 45%) when farmers’ dependence on groundwater increases.

Sourcing additional surface water to meet demand is impractical in the basin. First, the upstream areas of the Musi catchment are very dry. Second, surface water is already fully committed in the Musi Medium and Nagarjuna Sagar irrigation projects in the downstream region of the catchment. Third, increasing the current water transfers from Nagarjuna Sagar to Hyderabad to meet the agricultural demand of the Musi sub-basin is unlikely as this would severally affect current water uses in the Nagarjuna Sagar and Krishna delta systems.

In this context, a 13%-discrepancy in assessment of annual resource replenishment may have a hefty impact on the establishment of adaptive strategies for sustainable water use in the basin. As water-consuming crops such as paddy are largely dominant in the catchment (77% of the irrigated area), adapting the cropping pattern to annual water availability would be a promising response to consider, towards sustainable water management. Paddy and sugarcane are the highest water consuming crops in the Musi catchment followed by chillies, gram, sorghum, and vegetables.

In the model, an attempt was made to satisfy the water demand by switching area planted with paddy in summer and winter seasons to low water-consuming crops. The estimated annual unmet demand of 105 Mm³/yr of the “recharge yield” scenario was fulfilled by substituting about 7% of the paddy cultivation area whilst a further 3% were needed to balance the

additional 45 Mm³/yr of the annual unmet demand of the “discharge yield” scenario. Shifting from paddy cultivation to rainfed crops would enhance water productivity due to the lower production costs and higher returns per cubic meter of dry-crops (George et al. 2011b). However, in a context where water access is little constrained (but through access to a pump set) and cheap (subsidized electricity), optimizing water productivity is seldom an objective function of farmers who rather consider issues of access to market and food self sufficiency. Understanding farmers’ crop-choice dynamics in relation to water availability is crucial for designing sustainable water management strategies (for instance, Venot et al., 2010 on the case of Nagarjuna Sagar). Analysis of historical climate data of the Musi catchment shows that the variability of rainfall is increasing with decreasing trend of low rainfall events and a significant increase in extreme rainfall events (Deshpande and Kumar, 2012). Future climate projections are also showing increased rainfall variability as this century progresses which will impact the recharge rates and have serious implications for future groundwater availability (Patwardhan and Kulkarni, 2012). The need for effective tools ready to assess accurately long-term groundwater availability would therefore become even more crucial for sustainable water resource management and adaptation scenarios improvement. So, this small example showed how critical groundwater availability assessment can be for establishing recommendations for adaptive strategies and reinforces the need for using advanced estimation methods when possible.

6 Conclusion

The present paper proposed the development of a distributed numerical groundwater model on the Musi River sub-basin in order to help water management authorities to better integrate

groundwater resources assessments into their water allocation plans and strategies. First, the results argued for the successful implementation of a distributed numerical model at catchment scale for a hard-rock aquifer within a semi-arid basin. Second, one of the major benefits brought by the numerical model was the improved estimation of groundwater-resource sustainability in contexts where human interventions have seriously impacted water dynamics. By integrating complex interactions between components of the water budget in space and time, the groundwater model was able to provide a more comprehensive conceptual assessment of the groundwater-resource sustainability.

Based on 15 years of observation data, the numerical groundwater model showed a mean interannual deficit in the aquifer storage of $132 \text{ Mm}^3/\text{yr}$ which was in accordance with the observed long term decline of the water table. The groundwater-resource sustainability was defined as the mean volume of water that can be pumped for irrigation without causing any long-term change in groundwater storage (and, therefore, no decline in natural discharge rates) and was evaluated at $1017 \text{ Mm}^3/\text{yr}$ over the simulation period by the numerical model (discharge yield). This evaluation significantly differed (13%) from the standard mean interannual renewable recharge evaluated at $1170 \text{ Mm}^3/\text{yr}$ (recharge yield). Actual mean interannual extraction was evaluated to be $1235 \text{ Mm}^3/\text{yr}$. Based on the discharge yield value, a sustainable management plan would recommend to cut the actual mean interannual extraction by 18% to stabilise the drop, but only by 5% if based on recharge yield amount.

This had significant implications when groundwater sustainable yield assessment was integrated within a water-allocation modelling framework. The allocation modelling exercise showed that finding alternative water sources to sustain current water demand in the basin was hardly feasible. The only other option to reduce groundwater abstraction would be to cut the demand, which could be soundly achieved through changes in the cropping pattern. For

instance, when considering the 105 Mm³/yr-shortfall in water demand of the “recharge yield” scenario, 7% of the irrigated rice area had to be replaced by rainfed crops to put demand in line with long-term groundwater-resource sustainability. In the “discharge yield” scenario, 10% of the paddy area would be concerned for targeting the same objective (150 Mm³/yr-shortfall in water demand).

Using distributed numerical groundwater modelling when possible represents, therefore, a significant support for long-term sustainable water management strategies, based on a sound understanding of hydrological processes.

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Figures

Figure 1. General context of the Musi sub-basin, local features and groundwater situation. The color scale indicates the rate at which the water table (WT) has fallen over the period 1989-2004. OS: Osman Sagar reservoir, HS: Himayath Sagar reservoir; MR: Musi Medium Reservoir; NJS LC: Nagarjuna Sagar Left Canal

Figure 2. Evolution of the net irrigated area in the Musi sub-basin over the period 1989-2004 and relative proportion of the surface and groundwater sources (based on Handbook of Statistics, district wise data).

Figure 3. General groundwater level trend compared to the typical lithological profile. Evolution of the equivalent relative permeability (K_{global}) with depth (m below ground level) is sketched on the right hand side (after Dewandel et al., 2007). Permeability values from the calibrated model are reported in column Calib.

Figure 4. Groundwater model settings with observed water-table contours for the initial time step (May 1989).

Figure 5. Observed versus calculated annual piezometric heads. Heads of the first simulated year (1990) are black diamonds and heads of the following years (1991-2004) are grey diamonds.

Figure 6. Main components of the simulated annual groundwater budget for the period 1990-2004 and evolution of the model efficiency indicators (RMSE and abs.RM). Surface-water exchange and groundwater evaporation were too small to include as part of graphs. The calibration period is in light grey overlay.

Figure 7. Comparison of the relationships between rainfall and direct recharge calibrated in the model and inferred from the results of a study on granites and gneisses of South India that used a tritium tracer technique (Rangarajan and Athavale, 2000); r is the sample correlation coefficient.

Figure 8. Calibrated maximum (1998) and minimum (1992) annual groundwater net recharge values (R_{net}) and spatial distribution.

Figure 9. Comparison of estimated annual actual pumping, recharge yield and simulated discharge yield. Horizontal plain and dashed lines represent the interannual average of each component.

Figure 10. Conceptual diagram showing the simplified modelling framework. Outputs of the groundwater model (1: “recharge yield” scenario; 2: “discharge yield” scenario) are alternatively used as input to define groundwater availability in the water allocation model.

Tables

Table 1: Simulated mean interannual groundwater budget (Mm^3) over the period 1990-2004.

	R_{net}	E	River leakage	Baseflow	P	Lateral inflow	Lateral outflow	ΔS
Volume	1170	-16	<1	-23	-1235	4	-33	-132

Table 2: Mean interannual estimated values of the net recharge components (Mm^3) over the period 1990-2004.

	GW_{rf}	SW_{rf}	R_{rain}	R_{art}	R_{net}
Volume	370	30	648	122	1170
% P_{tot}	5	<1	9	2	17
% R_{net}	32	3	55	10	100

800 HJ-2012-2582 Français

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804 Amélioration de l'évaluation de la pérennité de la ressource en eau souterraine à l'aide d'une
805 modélisation déterministe : cas du sous-bassin semi aride Musi, Inde du Sud

806 RÉSUMÉ

807 Depuis les années 1990, des agriculteurs indiens, aidés par le gouvernement, sont
808 partiellement passés de l'irrigation par eau de surface à l'irrigation par eau de nappe en
809 réponse à la disponibilité incertaine de l'eau de surface. Les autorités gestionnaires de l'eau ne
810 commencent que lentement à considérer l'utilisation durable des ressources en eau souterraine
811 comme une préoccupation primordiale. Aujourd'hui, une intégration fiable des ressources en
812 eau souterraine dans un plan de répartition de l'eau est nécessaire pour prévenir une
813 surexploitation de l'aquifère. Sur les 11000 km² du sous-bassin de la rivière Musi (Sud de
814 l'Inde), les interventions humaines ont impacté de façon dramatique les aquifères du socle,
815 avec un abaissement de la surface libre de la nappe de 0.18 m/an sur la période 1989-2004.
816 Un modèle numérique de représentation de la nappe entièrement discrétisé a été utilisé avec
817 succès à l'échelle du bassin versant. Le modèle permet deux conceptualisations distinctes de
818 la disponibilité de l'eau de la nappe à quantifier : l'une est liée aux flux facilement quantifiés
819 l'autre exprime mieux la durabilité sur le long terme, grâce à la prise en compte de toutes les
820 sources et de toutes les pertes. Les simulations montrent que la dernière implique 13% d'eau
821 disponible de moins que la première. Corrélativement, ceci a une incidence majeure dans le
822 cadre de la modélisation de la répartition d'eau existant, utilisé pour guider les décideurs et les
823 gestionnaires des ressources en eau à travers le monde.

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827 HJ-2012-2582 Spanish

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829 Mejora de la evaluación de la sustentabilidad del recurso de agua subterránea con
830 un modelado determinístico: un caso de estudio de la subcuenca semiárida de Musi,
831 Sur de India

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836 RESUMEN

837 Desde los años 1990, los agricultores de India, apoyados por el gobierno, se han desplazado
838 parcialmente del riego con agua superficial al agua subterránea en respuesta a la
839 incertidumbre de la disponibilidad de agua superficial. Las autoridades de manejo del agua
840 solo lentamente comenzaron a considerar un uso sustentable de los recursos de agua
841 subterránea como una preocupación primaria. Actualmente, se necesita una integración
842 confiable de los recursos de agua subterránea en la planificación de la distribución del agua
843 para prevenir la sobreexplotación de acuíferos. Dentro de los 11,000-km² de la subcuenca del
844 Río Musi (sur de India), las intervenciones humanas han impactado dramáticamente los
845 acuíferos de rocas duras, con un descenso del nivel freático de 0.18 m/año durante el período
846 1989-2004. Se implementó exitosamente un modelo numérico de agua subterránea totalmente
847 distribuido en la escala de cuenca. El modelo permitió cuantificar dos conceptualizaciones
848 distintas de la disponibilidad de agua subterránea: una fue asociada a flujos fácilmente
849 cuantificables, y una que fue más expresiva de la sustentabilidad a largo plazo teniendo en
850 cuenta todas las fuentes y sumideros. Las simulaciones mostraron que esta última implicaba
851 13% menos agua subterránea disponible de para la explotación de lo que arrojó la primera. A
852 su vez, esto tiene grandes implicancias para la existencia del marco modelado de la
853 distribución del agua usado para guiar los que toman las decisiones y a los que gestionan los
854 recursos hídricos en todo el mundo.

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HJ-2012-2582 Português

Melhorando a avaliação da sustentabilidade dos recursos de água subterrânea por modelação determinística: um estudo de caso da sub-bacia semi-árida do Musi, Sul da Índia

RESUMO

Desde os anos 90 que os agricultores da Índia, apoiados pelo governo, alteraram parcialmente as suas origens de água destinada a rega, de águas superficiais para águas subterrâneas, em resposta à incerteza na disponibilidade de recursos hídricos superficiais. Só muito lentamente as autoridades de gestão da água começaram a considerar o uso sustentável dos recursos hídricos subterrâneos como a sua principal preocupação. A fim de evitar a sobre-exploração dos aquíferos, é necessária uma integração fiável dos recursos hídricos subterrâneos no planeamento e gestão da água. Numa área de 11,000 km², correspondente à extensão da sub-bacia do Rio Musi (Sul da Índia), as intervenções humanas têm afetado drasticamente os aquíferos de rochas fraturadas, com rebaixamentos da ordem de 0.18 m/ano durante o período 1989-2004. Foi implementado com sucesso, à escala de bacia, um modelo numérico de água subterrânea totalmente distribuído. O modelo permitiu simular duas conceções distintas de disponibilidade de água subterrânea para a sua quantificação: uma associada a fluxos facilmente quantificados e outra mais relacionada com a sustentabilidade a longo prazo, tendo em conta todas as fontes e sumidouros. As simulações mostraram que, nesta última conceção, 13% das águas subterrâneas estavam menos disponíveis para exploração do que na primeira. Este facto tem implicações importantes para a estrutura de modelação de alocação de água existente, a qual é usada para orientar as tomadas de decisão dos gestores de recursos hídricos em todo o mundo.

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