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Modelling the Effects of Groundwater-Based Urban Supply in Low-Permeability Aquifers: Application to the Madrid Aquifer, Spain

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Abstract The European Union Water Framework Directive establishes the obligation to all Member States to develop and implement catchment-scale management plans. These aim at reaching a good status of all water bodies in Member States by 2015 or, at the latest, by 2027. Numerical models provide a suitable approach to evaluate the possibilities of achieving this goal by enabling users to deal with complex hydrological interrelations in a dynamic manner. This paper presents a modellingbased approach to examine the past and future effects of groundwater-based urban supply in Metropolitan Madrid, Spain. Monthly scale model calibration is based on 32 years of water table observations. Care is taken to address the effect of intensive pumping on groundwater levels and streamflows, as well as the role of groundwater resources in meeting urban demands during dry spells. The paper concludes by reflecting on the implications and probabilities of meeting the aforementioned deadlines, as well as on the sustainability of past and future pumping trends. Policy considerations aside, modelling results suggest that groundwater extractions could be augmented considerably without causing a significant decrease in streamflows.

1 Introduction

Models have long since gained recognition as valuable tools to underpin decision making in the field of water resources (Moore 1979; Andreu et al. 1996; Koch and Grünewald 2009). This is partly explained by the ability of numerical approaches to deal with complex hydrological interrelations in a dynamic manner. Models however present other advantages. For one, models allow for comprehensive scenario evaluation, which provides the opportunity to carry out risk assessments and think about the future implications of management alternatives (Molina et al. 2009). Moreover, models are powerful communication devices, able to present results in an efficient and visually appealing fashion. This in turn implies that models can be used at most stages of the planning and management processes.

As simplified representations of reality, models require a clear conceptual definition of the physical system under consideration. Although models may be developed and used in an explorative way (Foppen 2002; Ramesh and Davison 2002; Bazzani et al. 2005), modellers must always strive to gain a sufficiently good understanding of the physical system before launching into the model development stage. Thus, the very process of elaborating a model may contribute to further the understanding of reality. Nevertheless, this also means that models are often subject to a plethora of uncertainties. Pitfalls to look out for include the use of models outside their validated scope, insufficient data and/or miscommunication between the modeller and the end-users (Refsgaard et al. 2005). Thus, results should always be handled with care, ensuring that the limitations of the modelling approach are clearly put forward. Another important consideration stems from the need to integrate the modelling and policy-making processes. This means that planners and stakeholders should ideally be involved from the very outset, contributing to define the modelling objectives and scenarios (Henriksen et al. 2007; Martínez-Santos et al. 2008, 2009).

Though often burdened by the absence of data, numerical models are particularly important to ensure an adequate use and protection of groundwater resources (Moore 1979; Alley and Emery 1986; Rainwater et al. 2003). This is largely due to the "hidden" nature of groundwater, which makes it a lot more difficult to track down and account for than surface water. Uncertainties typically affect aquifer recharge, groundwater abstraction patterns, the spatial distribution of hydrogeological parameters and the interactions between surface and groundwater bodies (Froukh 2003; Castaño et al. 2009; Manghi et al. 2009). Furthermore, optimal allocation of the available resources implies the need to address the potential effects of groundwater pumping in time and space (Zume and Tarhule 2008) A coherent integration of all such variables almost necessarily calls for numerical approaches. It is within this context that models provide one of the most adequate technological solutions available to deal with complexity (Martínez-Santos et al. 2005; Vázquez-Suñé et al. 2006; Sanz et al. 2009).

This paper presents a modelling-based approach to evaluate the past and future implications of groundwater-based urban supply in Metropolitan Madrid, Spain. In particular, this work focuses on examining the interaction between surface and groundwater at the sub-basin scale. The model was intended to support the elaboration of the Tagus basin catchment plan, whose development and implementation in turn responds to an explicit requirement from the EU Water Framework Directive (European Commission 2000). Once calibrated, the model was used to test policy scenarios devised by the Planning Office of the River Basin Authority.

2 Methods

2.1 Study Area

Madrid is located towards the middle sector of the Tagus basin, an 80,000 km^2 catchment shared by Spain and Portugal. The Tagus basin includes both Madrid and Lisbon, and is the most populated river basin in the Iberian peninsula. Approximately 80% of the population (5.5 million people) is concentrated within 50 km of Madrid, whose metropolitan region accounts for less than 10% of the system's areal extent (Fig. 1).
The region presents a continental, semiarid climate. Hot dry summers follow temperate winters; whereas long dry spells, lasting several years at a time, alternate with short wet periods. Rainfall amounts to 435 mm/year, taking place mostly in spring, autumn and winter (Fig. 2). Mean temperatures range from 6°C in January to 25°C in July, with the yearly average standing at 14.5°C (AEMET 2005).
The Madrid detrital aquifer is one of Spain's largest and most valuable groundwater systems. Freshwater reserves exceed 20,000 Mm^3, which makes the aquifer a strategic resource of the utmost importance, particularly during droughts (CHT 1997). A variety of studies have described the site from the geological, hydrogeological and hydrochemical perspectives (Llamas and Cruces 1976; López Vera 1976; Martínez-Alfaro 1977a; Villarroya 1977; Rebollo 1977; Sastre 1977; Llamas and Martínez-Alfaro 1981; Llamas et al. 1982; Fernández-Uría et al. 1985; MartinLoeches and Sastre 1993; Hernández-García et al. 1998; Hernandez-García and Custodio 2004; Samper et al. 2006). Thus, the aquifer essentially consists in a sedimentary basin contained within a 6,000 km^2 tectonic depression (Fig. 1). Abrupt mountains, impervious for practical purposes, bound permeable deposits from the southwest to the northeast. The topography gradually becomes gentler towards the
Tagus river, which runs along the southern end of the system. Rolling plains dominate a landscape that features a well-defined drainage network. Gaining streams cross the system from north to south, from the Central Range to the Tagus river. These were subject to strong seasonal fluctuations under undisturbed conditions. Over the last decades, however,

regulation through upstream dams attempts to maintain a suitable environmental flow. In practice this means that seasonal fluctuations are no longer as significant.

The Central Range consists mostly of schist and granitic outcrops. These dip sharply towards the SE, accounting for the geological basement of the groundwater system. Cretaceous limestones also outcrop towards the NE, giving rise to small, isolated aquifers of marginal importance. Erosional deposits of Tertiary age fill the tectonic depression. These change laterally from detritic to evaporitic facies. From NW to SE, sediments can be further classified in four sectors. Sediments near the Central Range are heterogeneous in size, ranging from silts and clays to boulders. These make up a low-permeability area towards the northwest of the system. A medium to low-permeability sector follows in SE direction. This is made up of sands and clays, which account for the larger share of the basin and constitute the main body of the aquifer. These gradually evolve to evaporitic deposits (gypsum), impervious for practical purposes. A transitional facies is observed in between, and consists in a very low-permeability mix of the two.

Permeable deposits exceed 3 km in depth, and are best described as a series of sand lenses embedded in a low-permeability clay matrix. From a geological standpoint these result from a classic superimposed alluvial fan configuration. Sand lenses typically run from NW to SE, and are several meters thick, tens of meters wide and hundreds of meters long. Some are interconnected directly or through transitional materials of intermediate granulometry. Sand gradually disappears with depth.

From a hydrogeological perspective sand lenses store water and drain the clay matrix, allowing moderate well yields for relatively short periods of time. On the other hand, clays account for the system’s hydraulic continuity. This implies that the system behaves as an aquitard, rather than as an aquifer (Toth 1995). Well yields depend on the number and size of the lenses encountered during the drilling process. For practical purposes this means that deeper wells are likely to encounter more sand, which largely explains why Madrid supply tubewells are several hundred meters deep.

The strategic importance of the aquifer is best understood from a historical perspective. Indeed, the very name of Madrid (*Mayrit*) means “city of *mayrats*” (i.e. “city of infiltration galleries”), and is derived from its ancient groundwater abstraction system (Llamas 1976). Infiltration galleries were sufficient to meet metropolitan water demands until the late 1850s, but began to struggle when the population reached

200,000. Faced with this water managers set out to imitate the Paris model, where a supply system based on deep wells had been successfully implemented. This intent failed for a variety of reasons, including the hydrogeological differences between the Madrid and Paris basins and the absence of sufficiently advanced drilling technologies. In contrast, an alternative supply system based on dams and canals successfully managed to bring good-quality surface water from the Central Range (Martínez-Alfaro 1977b).

Largely as a result groundwater supply was abandoned. Nevertheless, as population continued to increase the surface water system proved unable to guarantee supply during long droughts. In the 1960s water managers turned to groundwater for solutions. By then the advances in drilling and pumping techniques had already shown the potential of the aquifer as a complementary water source. Backup wellfields were thus drilled, becoming operational in the 1980s (Llamas et al. 1996).

In recent times, water demand amounts to approximately 500 Mm^3/year. About 95% of this is ordinarily supplied by 15 surface reservoirs, which present a joint storage capacity in the order of 950 Mm^3. The remaining 5% depends on groundwater. Usage of back-up well-fields augments this proportion to 25% during frequent dry spells, thus guaranteeing water availability (CYII 2003). Figure 3 plots rainfall cumulative deviations against groundwater extractions. Rainfall corresponds to the Madrid-Retiro station, located within the city of Madrid, whereas extractions account for both urban supply wells and private usage. A strong correlation exists between rainfall and groundwater pumping from the late 1980s. Since then, pumping is observed to exceed the average of historical extractions during dry years, but remains below in wet sequences. Widely oscillating patterns are explained by fluctuations in the extraction by urban supply wells, whose overall capacity exceeds the sum of all private users.

Piezometric records reveal a slight downward trend across most of the system (Fig. 4). A falling water table is largely an effect of groundwater pumping. Sharp drops are however attributed to measurement errors or dynamic drawdowns, as the trend is usually recovered soon after these take place. Observed drawdowns are most significant around supply well fields, particularly after long dry periods. Conversely, some piezometers show a remarkably stable evolution. This is mostly observed towards the southwest of the system, and can be explained by the historical absence of pumping wells in the area.

2.2 Model Development

From a conceptual standpoint it is generally accepted that rainfall is responsible for recharge of the Madrid aquifer. Infiltration takes place primarily in rainfall divides, whereas rivers constitute discharge outlets (Fig. 5). Evapotranspiration from the water table also accounts for a part of the system losses. Given the thickness and low permeability of the regional system, groundwater moves slowly and circulates deeply within. Regional flow conditions may be modified at the local level owing to a variety of reasons, which include the topography of the basin, its in-depth geometry and the anisotropic nature of aquifer materials. Stagnant areas may exist wherever equal magnitude vectors in the flow field cancel each other out. Groundwater residence time in the system typically amounts to several thousand years (Llamas and Martínez-Alfaro 1981; Llamas et al. 1982). This configuration responds to a conceptual model described by Hubbert (1940), Toth (1962, 1963) and Fetter (1994).

Since the 1970s, a variety of numerical approaches have attempted to replicate the behaviour of the aquifer. These consider both the regional and local scales. Llamas and Cruces (1976) used a finite difference multi-layer model to explain the conceptual performance of the system. A variety of modeling studies ensued. These not only focused on simulating aquifer behavior, but also recharge patterns and the likely evolution of the aquifer under different groundwater development assumptions (Lopez-Camacho and Lopez Garcia 1979; Llamas and Martínez-Alfaro 1981; Carrera and Neuman 1982; SGDGOH 1996). Local-scale models, on the other hand, focused on evaluating flow inside deep-supply wells (Samper et al. 1992), as well as on approaches to explain well-yield decay. More recently, Samper et al. (2006) coupled regional and local-scale approaches to assess the likely hydrogeological effects of building large urban tunnels along the Manzanares river. All such works significantly enhanced the understanding of the aquifer and its potential for water supply.

To an extent, the present model is a synthesis of the above. However, it also contributes to existing knowledge in several ways. For one, it significantly extends the calibration period in relation to previous works, which helps gain a better understanding as to how the physical system behaves over long periods of time. Furthermore, the model significantly enlarges the study area to the west. This is particularly important in as much as it allows for the incorporation of a new supply well-field, which is expected to provide up to 30 Mm^3/year during dry spells (Lopez-Camacho et al. 2006). The model also evaluates evapotranspiration from the water table and aquifer-river interactions, both from historical and future perspectives.

The model is implemented in Processing Modflow Pro (PMWin), a threedimensional groundwater flow and mass-transport modelling package (Chiang and Kinzelbach 2001; Chiang 2005). PMWin provides a graphical user interface for the classic finite-difference Modflow code (McDonald and Harbaugh 1988), whose reliability and robustness has been tested extensively. Modflow inputs include hydrological variables such as rainfall or pumping rates, as well as hydrogeological parameters like permeabilities and storage coefficients. Outputs are typically expressed in terms of water table elevations or stream discharges.

The aquifer is discretized in 109 columns and 99 rows, each cell accounting for an area of 1 × 1 km. Three layers are specified. This serves a twofold objective. First, it allows to reflect the telescopic geometry of the aquifer bottom, or, in other words, how the areal extent of the aquifer decreases with depth. In the second place, since PMWin assumes wells to be fully penetrating in each layer, a three-layer setting serves the purpose of establishing different depths for pumping wells. This is particularly important to reflect how increasingly deep supply wells have been drilled over time. Thus, the top of the upper layer corresponds to the topographic surface of the system, whereas its bottom is defined at 350 m.a.s.l. This caters for the first generation of supply wells. The second layer is 200 m thick, in order to incorporate the 700 m-deep wells that currently supply the metropolitan area of Madrid. Finally, the third layer represents the aquifer depths yet to be exploited, reaching down to 2,000 m below sea level (Cadavid 1977).

Approximately 60% of the model domain corresponds to active cells. These are bound by inactive areas, which in turn account for the hydrogeological limits of the system. No-flow boundaries are prescribed to the north and south. These cater for the impervious materials of the Central Range and the transitional sedimentary facies. This boundary condition also applies to the southwestern end of the grid, which represents a short basin divide separating the Alberche and Tagus rivers. River cells, corresponding to the Jarama and Alberche streams, make up the eastern and western limits of the model.

River cells are also specified for major streams. To account for exchange flows between the aquifer and the stream at each cell Modflow uses a variety of parameters. These include hydraulic conductivity of the riverbed (m/d), river stage (m), elevation of the riverbed bottom (m), width of the river (m), length of the river within the cell (m), and the thickness of the riverbed (m). All except for stream stage are grouped under a single parameter, namely river conductance. This accounts for the material and characteristics of the riverbed and its immediate environment.

Though often omitted in groundwater modeling studies, direct evapotranspiration from the water table may play a significant role in aquifer water balances (MartínezCortina and Cruces 2005). In this case, evapotranspiration cells are prescribed along discharge areas (valley bottoms) in order to simulate the effects of plant transpiration and direct evaporation from the saturated zone.

For modeling purposes Modflow's evapotranspiration package uses three parameters, namely the maximum evapotranspiration rate (m/d), the elevation of the evapotranspiration surface (m), and the extinction depth (m). In each stress period water is removed from cells depending on the water table elevation. If the water table is at or above the elevation of the evapotranspiration surface, the model takes away water using the maximum allowed rate. Conversely, no water will be removed whenever the water table remains below the extinction depth. The evapotranspiration rate is assumed to vary linearly in between. For the purpose of this model the evapotranspiration surface was assumed to coincide with the topographic elevation, whereas maximum rates were defined according potential evapotranspiration records. Average evapotranspiration in the region amounts to 750 mm/year, oscillating from 10 mm in January to over 140 mm in July (CHT 2002). Several average extinction depths were tested during the calibration process, ranging from 1 to 3 m. A sensitivity analysis showed the effects to be negligible.

Recharge is understood as net infiltration reaching the water table, and is assumed constant during each stress period. For modelling purposes recharge was assumed uniform across the aquifer.

Recharge is perhaps the single most difficult element to quantify. Causes should be found in the low vertical permeability of the system, as well as in the thickness of the unsaturated zone in recharge areas. Both imply that recent infiltration may take thousands of years to reach the water table. Consequently, it makes relatively little sense to account for yearly scale variations in rainfall. Most existing studies work with average values. For instance, based on water balances Martínez-Alfaro (1977a) and Rebollo (1977) estimated regional-scale recharge in the Manzanares and Guadarrama sub-basins to be 40–50 mm/year (i.e. about 8.5% of the average rainfall). These estimates have been later confirmed by other sources (CYII 2003).

In practice, local-scale hydraulic conductivity is subject to dramatic changes. Given the scale of the model, some regionalization of hydrodynamic parameters was needed. Various parametrization approaches of heterogeneous media have been proposed over the years, most of which assume an equivalent homogeneous medium. Parametrization techniques often require extensive fieldwork, particularly in large aquifer systems characterized by heterogeneity. Carrying out pumping tests under sufficiently adequate conditions might not always be possible, which sometimes results in the need to extrapolate few results to very large areas. Methodologies based on more accessible data, namely specific capacity, are thus perceived as welcome alternatives to deal with heterogeneous systems (Martínez-Alfaro and LopezCamacho 1979).

In this case, definition of hydrogeological parameters, namely permeability and specific yield, is done in agreement with results from over 50 pumping tests. Correlation of specific capacity and transmissivity allowed to obtain permeability estimates for a further 300 boreholes.

Specific capacity is defined as the ratio between the pumping rate and the induced drawdown or, more specifically, as yield per unit drawdown. This relationship must be measured over a sufficiently long time so that drawdowns can be assumed stable. Specific capacity not only reflects hydrogeological parameters but also borehole construction factors. Take for instance well radius: a larger radius increases the grid area of the well, and hence the volume of water that can flow in. Consequently, a larger radius allows to obtain the same yield for a smaller drawdown.

Pumping test interpretation by Theis method yields 56 “real” transmissivity values. A Gumbel cumulative probability graph is then plotted, obtaining a 95% adjustment by the chi-square test. Transmissivity values are observed to fall within a log-normal distribution whose geometric mean (i.e. its median) equals 10.5 m^2/d. Specific capacities are in turn calculated considering drawdowns after a 24-h pumping period. Specific capacity data follows a similar log-normal law, with an average of 0.5 l/s/m and a median of 0.25 l/s/m.

A strong correlation is observed between pumping test transmissivities and specific capacity. Thus, the relation between transmissivity and specific capacity is established according to Eq. 1:

$$T = 0.64q + 2.21 \quad (1)$$

Where T represents transmissivity (m^2/d) and q is the specific capacity (l/s/m) multiplied by a factor of 100. This expression readily allows to derive transmissivities from specific capacity data. For the purpose of the model, permeabilities can in turn be inferred from transmissivities.

A spatial overview of the above parameters suggests that the system can be divided in three roughly homogeneous zones (Martínez-Alfaro and Lopez-Camacho 1979). These respond to the regional sedimentation model. Permeability thus varies from the Central Range to the Tagus valley, following the depositional sequence of sediments. Sedimentation areas near the mountains feature significant heterogeneities, as large boulders (several metres in diameter) alternate with finer materials such as clays. Hence the average hydraulic conductivity is low (about 0.01 m/d). Increasing grain homogeneity yields higher permeability values as the distance to the mountains increases (0.25 m/d). This corresponds to the central area of the model. Finally, permeability becomes lower near the transitional facies (0.1 m/d), which in turn represent the southern border of the system. Overall, the anisotropy ratio (horizontal/vertical) ranges between 100 and 1,000.

Specific yield, also known as drainable porosity, is defined as the amount of water that can be extracted per unit area of a rock body per unit decline in the groundwater level. Since this water occupies void spaces in the rock, specific yield can be expressed as a percentage of the total rock volume. Specific yield only refers to the water that can be drained by gravity, i.e. it does not take into account the thin film of water that adheres to the rock, and is usually obtained by means of pumping tests. Specific yields in the area range between 7% and 10% (SGDGOH 1996; Samper et al. 2006).

2.3 Model Calibration

Model calibration is carried out in two stages, namely steady and transient-state conditions. A steady-state calibration run was carried out first to reproduce aquifer conditions prior to intensive pumping. These are in turn used as the starting point for the transient-state historical calibration. Flow equations were solved using the strongly implicit procedure (Chiang and Kinzelbach 2001). Under steady-state conditions, groundwater levels calculated by the model are largely dependent on the ratio between recharge and permeability. As discussed earlier, both these parameters were assumed "known" to an acceptable extent based on existing studies. Recharge is thus taken as a lumped estimate and distributed evenly across the system. On the other hand, permeability is regionalized in three different sectors (north, central, south) based on field data.

Calculated steady-state heads were validated against the 1975 water table map (Martínez-Alfaro 1977a). This is considered to be sufficiently representative the natural conditions of the system because groundwater pumping did not become widespread until the late 1970s. Figure 6 presents the calculated water table levels for quasi-natural conditions. As expected, these showcase a north-to-south regional flow, as well as a gaining behaviour on the part of the main streams. Transient-state calibration spans a 32-year interval (1975–2007), comprising 384 monthly stress periods. These depart from the steady-state results, thus allowing to evaluate the evolution of the groundwater system from quasi-natural conditions to current times.

All transient runs include pumping as a system output. Pumping data was provided by the Planning Office of the Tagus River Basin Authority, and distinguishes between groundwater supply wells and private groundwater usage. Records corresponding to supply wells are reliable, but data for private extractions varies in quality. Overall, the 1975–1996 records are more complete. Due to the absence of comprehensive private abstraction records for the remainder of the series (1996–2006), the existing information was complemented with a field survey and through the interpretation of satellite images. These were conducted by a private company, and mostly focused on identifying groundwater consumption in golf courses, irrigated agriculture and newly built urban nuclei.

Inverse calibration is often used to estimate uncertain parameters in modeling studies (Chiang 2005). This is achieved through simultaneous parameter-fitting runs, which ultimately aim at establishing a good agreement between calculated levels and a long dataset of observed hydraulic heads (i.e. minimizing the sum of squared deviations). Long observation records enhance the reliability of the model, particularly if these correspond to a spatially representative sample of monitoring wells. Some pitfalls should however be avoided. The strong interrelation between parameters such as recharge rates and hydraulic conductivity may sometimes lead to spurious outcomes (Sanford 2002).This can be overcome by knowing the magnitude and spatial distribution of hydrogeological parameters to a reasonable extent.

The monthly scale model was calibrated using PMWin’s PEST module (Doherty et al. 1994). Initial guesses for each of the hydrogeological parameters were obtained from pumping test data as described earlier on. These were adjusted simultaneously to match observation datasets from 28 monitoring wells. Care was taken to ensure that each piezometric series spanned a sufficiently long period of time (i.e. at least 20 years). Overall, the average number of field measurement per observation well exceeds 220. Observation wells were widely distributed across the system.

Parameter adjustment yields optimal permeability values of 0.10, 0.27 and 0.09 m/d for the northern, central and southern hydrogeological sectors respectively. Bestfit specific yields amount to between 7 and 10%. For these values the Pearson R^2 correlation coefficient between observed and calculated groundwater heads ranges from 0.865 to 0.921. Correlation coefficients are computed from a sample of 6,175 data pairs.

Discrepancies can be attributed to the heterogenous nature of the system, which may induce variations at the observation well scale. These could not be accounted for by using large equivalent-permeability zones. In addition, 1996–2007 pumping data

seemingly underestimates the actual magnitude of groundwater extractions in some areas (M. Casado, Tagus Basin Authority Planning Office, personal communication, October 2007). Uncertainties stem from the interpretation of satellite images, as well as from data obtained during the field campaign. These in turn provide a plausible explanation as to why the calculated trend gradually becomes more detached from the observed levels in some piezometers from the mid-1990s (Fig. 7). In any case, the model shows an acceptable agreement with field observations for a sufficiently long period of time (i.e. over 20 years). It is thus considered calibrated for practical purposes.

2.4 Simulations

Four forecast runs were conducted in this case. All of them consider a double horizon, 2015 and 2027. The choice of two simulation horizons is in compliance with the deadlines established by the WFD. The WFD establishes the obligation to reach a good ecological status of surface and groundwater bodies by 2015 or, after two 6-year extension periods, by 2027 (European Commission 2000). Simulations begin from the 2007 water table map, drawn up from field observations, and adopt yearly time steps. Water table drawdowns and river baseflows are used to compare the relative performance of each run. Pumping patterns for the four forecast simulations are presented in Fig. 8.

A planning officer from the TRBA was involved in developing and testing scenarios. The ultimate goal of the simulations is to provide a reasonably broad envelope of plausible management scenarios. Since evaluating all possible alternatives is unrealistic, scenarios have been designed to combine extreme conditions with more likely trends. In practice, this means that the likely outcome of any alternative scenario can be interpolated from those considered here.

A business-as-usual simulation is carried out first. This aims at establishing the likely evolution of the system if current conditions remain applicable throughout the next 20 years. Best and worst-case conditions are then established. "Best" and "worst" refer to the projected state of the aquifer and its associated surface water bodies. Thus, the best-case scenario is that which renders a hydrological condition that approaches the good ecological status (i.e. the natural state of the aquifer). Conversely, the worst-case scenario is that which results in a severely depleted water table. A safe-yield scenario completes the others. All alternatives are described below.

2.4.1 Scenario A

This first scenario corresponds to business-as-usual conditions, and establishes a trend with which to compare the results from other simulations.

All scenarios distinguish between private and urban supply extractions. Scenario A thus considers private pumping to remain roughly constant over time (50 Mm^3). This figure actually doubles the estimation for the 1996–2006 interval. As explained earlier, this is because such figures were found too low by TRBA personnel. In terms of urban-supply extractions, the 1996–2007 pumping sequence is assumed to take place over and over again. Further supply extractions are however added, taking into account the influence of well-fields currently under construction. These in turn allow for population growth forecasts. Overall extractions for this scenario range between 55 and 140 Mm^3/year.

2.4.2 Scenario B

Given the importance of groundwater supplies during dry spells, this simulation examines the performance of the aquifer system under intensive drought-related exploitation. This was initially considered the worst-case scenario from the management perspective.

Hydrological drought is defined as a significant decrease in the availability of water in all its forms appearing in the land phase of the hydrological cycle (Nalbantis and Tsakiris 2009). Various conceptualizations exist to compute drought indexes (Vicente-Serrano 2006; Tsakiris et al. 2006), a description of which falls outside the scope of this paper.

For modelling purposes, the likelihood of encountering droughts in future years is obtained from Hernandez-García and Llamas (1995). Based on 135 years of rainfall data, these authors established that approximately half of the hydrological years in the Madrid basin fall under the description of "dry" (i.e. rainfall falls short of the historical average by more than 15%). To them, droughts are understood as multi-year dry periods, which usually last from 2 to 4 years. The longest drought on record spanned 10 years (1866/1867–1875/1876).

In order to replicate similar conditions, scenario B assumes a severe 10year drought. Extractions from the aquifer during this period are established at 150 Mm^3/year, a figure which exceeds the maximum yearly extraction on record. For simulation purposes, two-thirds of this value are attributed to urban supply, whereas the remainder corresponds to private groundwater users. Urban-supply pumping stops in 2017 so as to evaluate aquifer recovery. Pumping by private groundwater users is maintained until the end of the simulation period.

Drought is assumed not to have an effect on annual aquifer recharge. This is because of the combined effect of a low vertical permeability and the thickness of the unsaturated zone in recharge areas, which in practice mean that infiltration may take hundreds of years to reach the water table. This assumption is consistent with water table observations, which show the aquifer to be largely indifferent to rainfall fluctuations.

2.4.3 Scenario C

Scenario C corresponds to a "safe-yield" simulation. In other words, extractions equal recharge throughout the modelling period. The 2006 spatial distribution of pumps is assumed to hold throughout. Extraction values for groundwater supply wells and private pumps are adjusted by multiplying the 2006 figure by a suitable factor. As discussed later, this scenario turned out to yield worst-case hydrological conditions by the end of the simulation period.

2.4.4 Scenario D

Finally, a zero-pumping scenario is considered. Although unrealistic for practical purposes, this estimates how long it would take to restore baseflows in streams and rivers under optimal conditions. As discussed later, this ultimately serves the purpose of evaluating whether WFD deadlines could be met.

3 Results

3.1 Historical Simulation

The historical simulation corresponds to the calibration run that provides the best adjustment between observed and calculated levels. Ensuing interpretations evaluate the evolution of the water table, as well as changes in streamflows. The water table map and aquifer discharges obtained for steady-state conditions are used as benchmarks for comparison.

Thus, maximum drawdowns after the 32-year period correspond to groundwater supply well-fields, where induced depths exceed 50 m. Conversely, little effect is observed towards the western area of the system. This is explained by two concurrent circumstances. The first—and most obvious—is that pumping is largely absent in the area. The second is the low permeability of the system, which typically results in pumping cones which are both narrow and deep. Consequently, the effects of intensive pumping may take many decades before they are felt in distant points of the aquifer.

Figure 9 shows the historical evolution of aquifer discharges against the aquifer's depletion curve. Total storage depletion for the 1975–2006 interval exceeds 1,200 Mm^3. Despite long-term storage deficits, rivers are observed to maintain a gaining regime. Baseflows have however diminished by approximately 15% (Table 1). This fact is strongly linked to a decline in direct evapotranspiration from the water table. Ultimately, both trends are a consequence of groundwater pumping.

A breakdown of historical river baseflows as calculated by the model is presented in Fig. 10. Trend lines represent aquifer discharges through the area's major streams (Alberche, Guadarrama, Manzanares and Jarama), including their tributaries. Results for the Alberche and Jarama rivers only account for one margin, since both rivers were taken as model boundaries.

Table 2 shows key statistical parameters for each one of the major streams. The Manzanares stream, central to the system, experiences the clearest downturn from quasi-natural conditions. Current baseflows only amount to 67% of the original aquifer discharge. This can largely be attributed to the nearby presence of important supply well fields. The other three major streams present comparatively smaller reductions. The Alberche stream, whose catchment is largely well-free, remains closest to quasi-natural conditions (90.1%).

Baseflow results may provide a cross-checking mechanism to validate the headwise calibration of the model. This is however contingent on the availability of baseflow data for the river reaches under consideration. In the case at hand, it is generally accepted that baseflows only provide a small share of total streamflows. As shown by Tables 3 and 4, this implies that the model is qualitatively correct in the assessment of groundwater discharges through streambeds. Indeed, streamflows in the area have long since been observed to depend heavily on the amount of water released from the dams in the Central Range, rather than on groundwater discharges. However, there are very few independent estimates of baseflows that can be used to evaluate the accuracy of the model. These in turn refer to short river segments and are only available for short periods of time. Hence they only have a partial value as calibration elements.

Take for instance the middle sector of the Manzanares river, where there are two stream gauges (263 "El Pardo" and 70 "Parque Sindical") separated by approximately 5 km. An attempt to evaluate the magnitude of baseflows between both stations was carried out in the summer months of 1991—July, August and September. The Water Authority estimated an average baseflow in the order of 308 l/s for the whole sector (CHT 1991). This is roughly consistent with modeling results, which render a discharge value in the order of 299 l/s for that same sector and period of time.

3.2 Forecast Runs

Figure 11 presents overall baseflow trends for all four simulated scenarios. Total baseflows, i.e. the sum of baseflows in all four main streams, are plotted. The calibrated historical trend is included for reference. As shown, despite the breadth of alternatives, differential effects on baseflows are relatively small. Indeed, rivers retain their gaining behaviour even under worst-case scenario conditions. Total discharges range between 100 and 115 Mm^3/year by the end of the simulated interval (80% to 92% of the natural baseflow).

Overall, Scenarios A, B and C result in a further decrease in total baseflows, whereas Scenario D is the only one to trigger a recovery in the system. Scenarios A and B render a similar reduction in overall baseflows by 2027. Curve shapes are however constrained by different pumping patterns. Scenario B yields a more noticeable decrease for the first 10 years. This is attributable to drought-related extractions. As these stop, however, discharges slowly stabilize. A marked drop is observed for scenario C, where no stabilization occurs during the modelled period. The implications are discussed later on. Finally, scenario D presents a fairly rapid upturn for the first 2 years, which is soon replaced by a more gradual recovery rate. This particular effect is largely a consequence of a rising water table in discharge areas, which in turn results in steadily increasing evapotranspiration from the water table. Best-case scenario conditions fail to recover the baseline stream discharges (125 Mm^3) by the end of the simulation period.

Table 3 presents a breakdown of projected flows per subcatchment, hence allowing to evaluate how the scenario-related reduction in baseflows could be felt at the basin scale. The Alberche stream remains remarkably stable throughout all simulations, whereas the Manzanares is the most vulnerable. Causes should be found in the absence of wells near the former, which remains safe from water table drawdowns for the most part, whereas the latter is located well within the main pumping area. Scenario C renders the worst baseflow conditions overall, with the Manzanares stream dropping to as little as 45% of the natural baseflows. On the other hand, scenario D would trigger significant recoveries in all cases. Modelling results yield baseflows in excess of 90% of the original value for three out of the four streams.

Simulated conditions influence the spatial distribution of the water table in a noticeable manner. By the end of the simulation period, scenarios A, B and C present considerably developed pumping cones (Fig. 12). These correspond to wellfield areas in all instances. In the case of scenario B, these remain well established even after groundwater supply pumps have been at a stop for 10 years. Drawdowns in excess of 60 m are observed in the pumping areas near Madrid. These are considerably more noticeable in scenario C, where sizeable 100 m pumping cones are observed. On the other hand, scenario D yields an improved water table condition compared to 2007. More specifically, pumping cones have largely disappeared and water table contours are observed to be in the process of recovering their natural shape. However, they still remain noticeably different to 1975 levels.

Since Scenario D considers a drastic reduction in pumping patterns, model results suggest that the aquifer is likely to remain in heavily modified conditions in the long term. Nevertheless, as discussed later the aquifer presents remarkable buffering capacity. According to the model, a dramatic increase in groundwater pumping (scenario C) would have a noticeable effect on baseflows. This, however, would only amount to a small share of total streamflows, since upstream dams account for most of the flow.

4 Discussion

The European Union Water Framework Directive (WFD) establishes the obligation to all Member States to restore a good ecological status of surface and groundwater bodies (European Commission 2000). This is increasingly becoming an issue of concern for water managers and practitioners across the EU, since the 2015 deadline draws nearer and the transposition of the WFD into national legislation is yet to be fully implemented in some countries. Partly as a result, forecasts from degraded basins suggest that many of the Union's water bodies are at risk of not complying with the WFD objectives (Sanchez 2003; Martínez-Santos et al. 2008). Cases are reported where there still may be a need for further monitoring in order to assess the likelihood of meeting the environmental goals (Karaczun 2005; Dworak et al. 2005; Laszlo et al. 2007), whereas a debate still exists as to what is meant by good ecological status (Borja and Elliott 2007). There are also instances in which the implementation of the Directive may meet difficulties due to institutional constrains or to existing social and economic developments—most notably those related to agricultural activities. Some authors even question the feasibility of compliance in the face of global risks such as climate change (Wilby et al. 2006).
WFD provisions in regard to groundwater resources are generally perceived as beneficial, although a variety of valid criticisms have been also put forward. These often challenge the definition of the quantitative status of groundwater bodies as a function of stream discharges and groundwater levels. While this could be appropriate in humid regions, the good quantitative status of aquifer-river systems is extremely difficult to assess in semiarid settings. In this regard, the WFD establishes that extractions should not exceed the available groundwater resources, which are defined as "the long-term annual average rate of overall recharge of the body of groundwater minus the long-term annual rate of flow required to achieve the ecological quality objectives for associated surface waters, to avoid any significant damage to associated terrestrial ecosystems". This leaves two major uncertainties, namely aquifer recharge and suitable baseflows. These are both extremely difficult to quantify due to the huge natural variability of hydrological systems in semiarid regions, as well as to the regional significance of unsaturated-zone processes. Some authors contend that a strict interpretation

of this definition leads to conclude that no groundwater resources are ever available (Samper 2005).

In the case of urban groundwater resources, central focus to the paper, the possibility of complying with these criteria is further constrained by the need to maintain suitable drinking water supplies to large metropolitan areas. As shown by modelling results, it is unlikely that natural conditions in the Madrid aquifer could be restored within the established deadlines, even if groundwater pumping came to a total stop. Therefore, it could be argued that it is technically unfeasible to recover the system before 2027. Moreover, the cost of recovering the aquifer would probably be disproportionate, as climatic constrains, namely droughts, render it a key strategic resource. These issues, together with the heavily modified condition of the surface and groundwater systems, imply that less stringent goals should apply.

According to modelling results, no plausible alternative is likely to recover the system. Scenarios A, B and C all suggest that baseflows will continue to decrease. This is in itself suboptimal, although the environmental implications need to be put in perspective. Obviously, there is no such thing as "no impact" when water is abstracted from nature, because there will always be a critical abstraction rate (related to both groundwater recharge and influence of river discharge regime) above which groundwater levels will fall, groundwater quality will deteriorate, and/or groundwater flows to rivers and wetlands will be reduced to a level that threatens ecosystem health (Lerat 2005). In this case, however, the contribution of aquifer discharge to total streamflows is relatively small even in natural conditions. Besides, prior to pumping river flows were already dependent on releases from upstream dams to a large extent (Tables 2 and 4). For practical purposes, this may have important management implications as to the effect of mounting groundwater extractions on streamflows.

Take for instance scenario C. This can be considered an extreme case in the sense that yearly extractions nearly triplicate the average of the 1996-2006 decade (140 Mm^3/year against 54 Mm^3/year). Moreover, extractions in scenario C are sustained over 20 years, while, in real life, the aquifer is allowed to recover during wet periods (when urban demands are met by surface reservoirs). A combination of historical records and model results show that baseflows amount to 10.7%, 19.5%, 2.8% and 1.2% of yearly streamflows in the Alberche, Guadarrama, Manzanares and Jarama streams, respectively (Table 3). On the other hand, scenario C predicts baseflow reductions of 0.1%, 1.8%, 22.8% and 19.1%. The product of these values yields a

total streamflow decrease of 0.01% for Alberche, 0.35% for Guadarrama,
0.63% for Manzanares and 0.23% for Jarama by the year 2027. Furthermore, the model renders no signs of aquifer exhaustion. Together with the above reasoning, this suggests that the aquifer would be able to withstand a considerable increase in groundwater extractions in exchange for a relatively negligible effect on surface water bodies.

This highlights the role of the low-permeability aquifer materials. In predominantly clayey environments such as the one at hand, pumping cones are typically well-developed in the vertical direction, but limited in horizontal extension. The low diffusivity of the system implies that perturbations, i.e. pumping, can take a long time to propagate and be felt at long distances from the origin. As a result, rivers may remain relatively safe from the effects of intensive pumping for very long periods of time. In other words, the low-permeability of the system provides a buffer in terms of how groundwater extractions affect surface water bodies.

Buffering capacity can also be approached from the sheer size of the system, which allows it to cope with heavy pumping during long droughts in exchange for a relatively minor environmental impact. Modelling results thus suggest that the aquifer could respond satisfactorily in the face of mounting demands.

From a different perspective, scenario C brings about somewhat of a paradox. “Safe-yield” conditions actually render the most undesirable result of all (Fig. 11). Of course, this is related to the fact that equalling extractions to recharge implies a threefold increase of pumping. However, it also raises a conceptual point, thus leading to the long-standing debate as to what kind of benchmarks should be used to evaluate sustainable development (Price 2002; Rogers et al. 2006; Llamas et al. 2006).

Flawed concepts such as safe-yield or aquifer overexploitation are often used to report whether aquifer resources are being extracted at a sustainable rate (Custodio 2002). Attaining safe-yield is generally understood as reaching a balance between groundwater extractions and natural replenishment rate of an aquifer over a long period of time. The underlying logic is that no change in storage will take place if a volume of groundwater equivalent to the natural recharge is abstracted. In turn, aquifers are often considered over-exploited if extractions exceed replenishment.

One of the main conceptual errors behind this reasoning is that it ignores the interactions between surface and groundwater bodies. Under natural conditions, a long-term equilibrium is reached between aquifer inflows and outflows. Thus, aquifer replenishment is balanced by discharges through springs, wetlands and streams. Pumping a volume of groundwater equal to aquifer recharge offsets discharges, drying up surface water bodies as a result (Sophocleous 1997, 2000). Hence, safeyield leads to the loss of aquatic ecosystems. Though the case at hand is not dramatic, model results are coherent with this interpretation. Streamflows are observed to decrease even though pumping equals recharge.

Groundwater levels are also taken as a benchmark to evaluate the sustainability of pumping trends. Downward water table records are often taken as a clear indicator of unsustainable groundwater use. Nevertheless, this is not always true. Falling levels may simply account for well interferences, or for the transient lag that follows pumping-induced changes in the balance between inflows and outflows (Custodio 2002). Since this may take thousands of years, the available data appears insufficient to provide a definite judgement. Overall, most observation wells show a downward drift between 1970 and 2005. Nevertheless, many of such piezometric trends have roughly stabilized over the last 10–15 years, and some even show a slow recovery
(Fig. 4).

By the very nature of the model, the above analysis only considers water quantity and its conclusions are therefore contingent of other issues. These include groundwater quality considerations, as well as the social and economic implications of drilling deeper wells as the water table drops and pumping cones get wider. While these aspects are acknowledged to be potentially important, they are considered to fall outside the scope of this work.

5 Conclusions

This paper has presented a modelling-based approach to examine the past and future effects of groundwater-based urban supply on the Madrid detrital aquifer, Spain. A monthly scale groundwater model was developed and calibrated based on 32 years of piezometric records. The calibrated model shows that streamflows partially depend on groundwater levels. It also replicates how sustained pumping over the last decades has significantly depleted the water table at the local scale, altering aquifer discharge into surface water bodies. Conversely, all streams are observed to maintain a gaining regime throughout the both the historical and simulation runs. Management scenarios were devised and modelled together with a representative of the Tagus River Basin Authority.

Simulations suggest that meeting the optimal objectives set by the Water Framework Directive is highly unlikely. This is both due to hydrological and socioeconomic constrains. On the other hand, aquifer freshwater reserves remain huge in comparison to streamflows and extraction rates. Model results imply that even a considerable long-term increase in extractions would not induce significant changes in streamflows.

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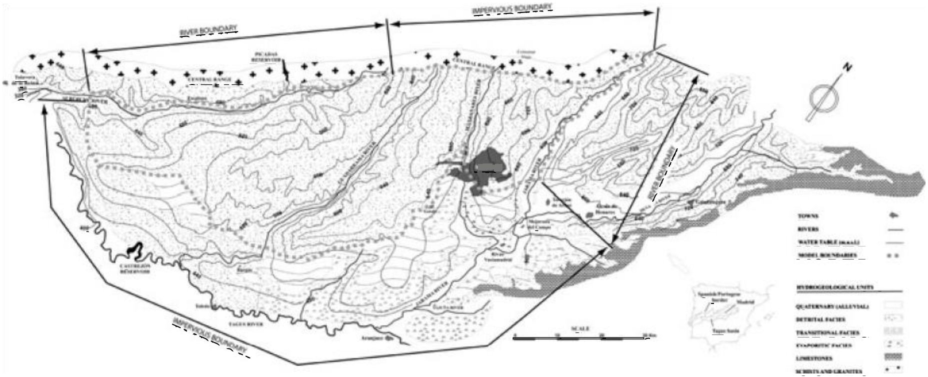
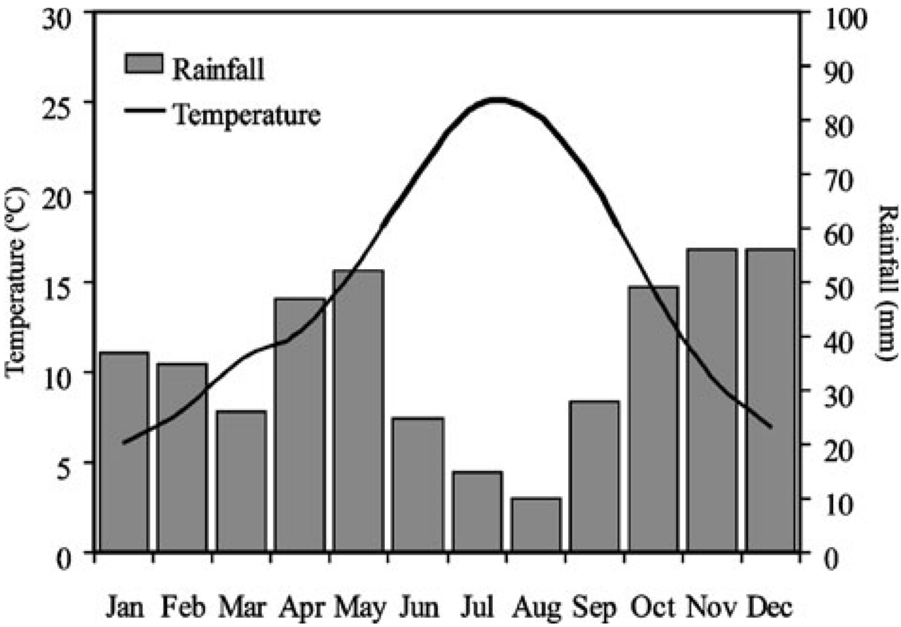


Fig. 1 Geographical and geological setting



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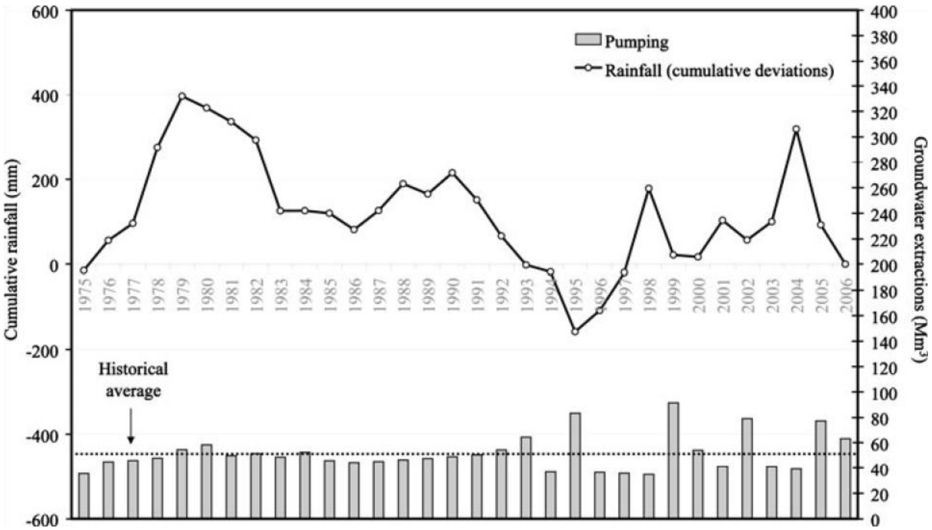

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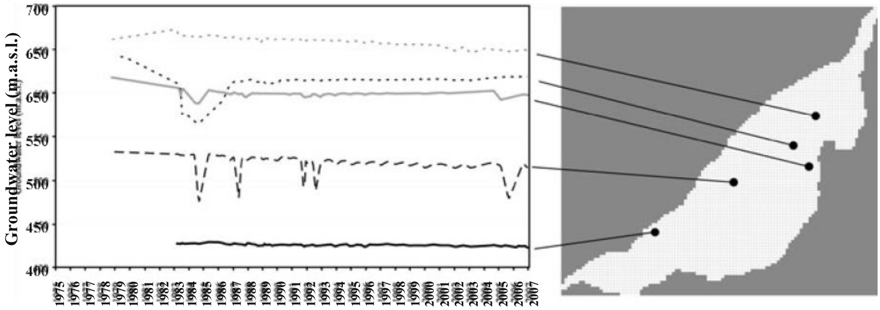

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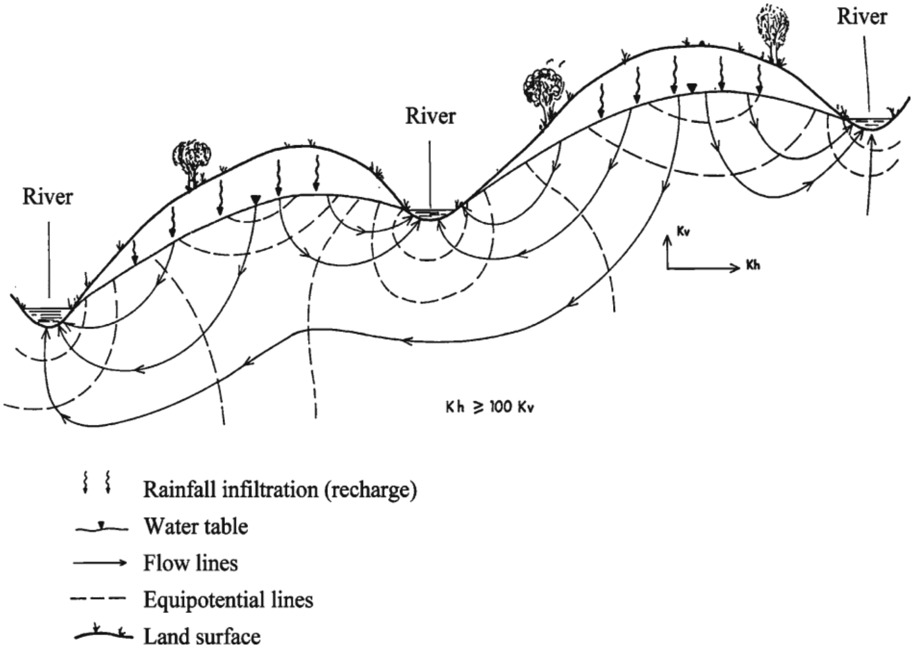

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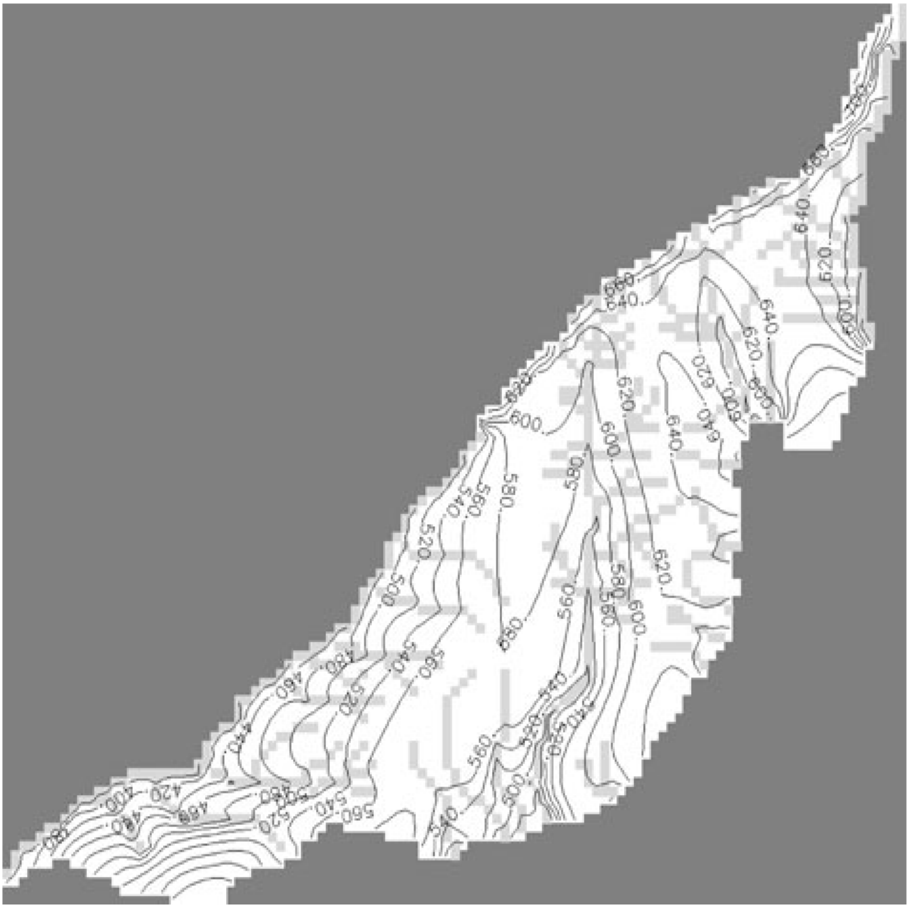
Accepted ms, Water Resour Manage (2010) doi: 10.1007/s11269-010-9682-0



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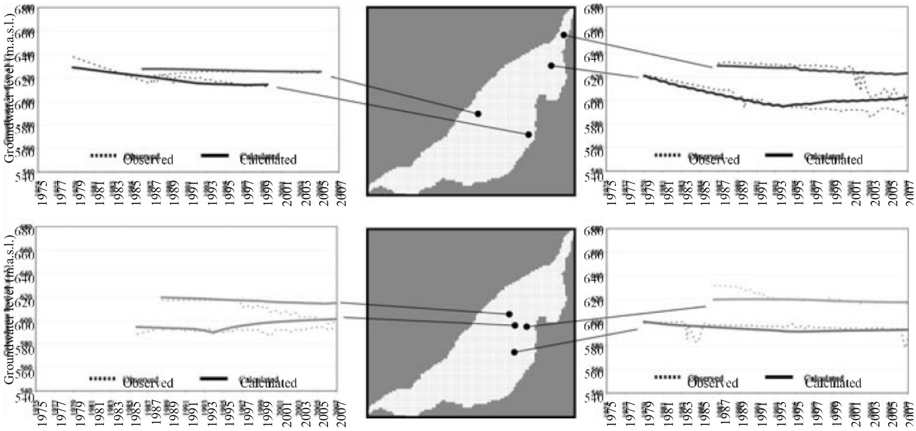

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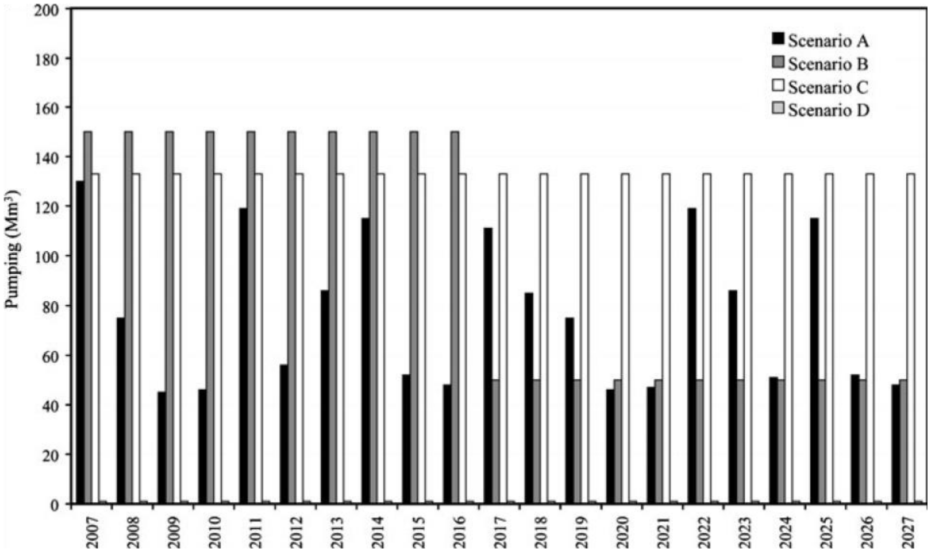
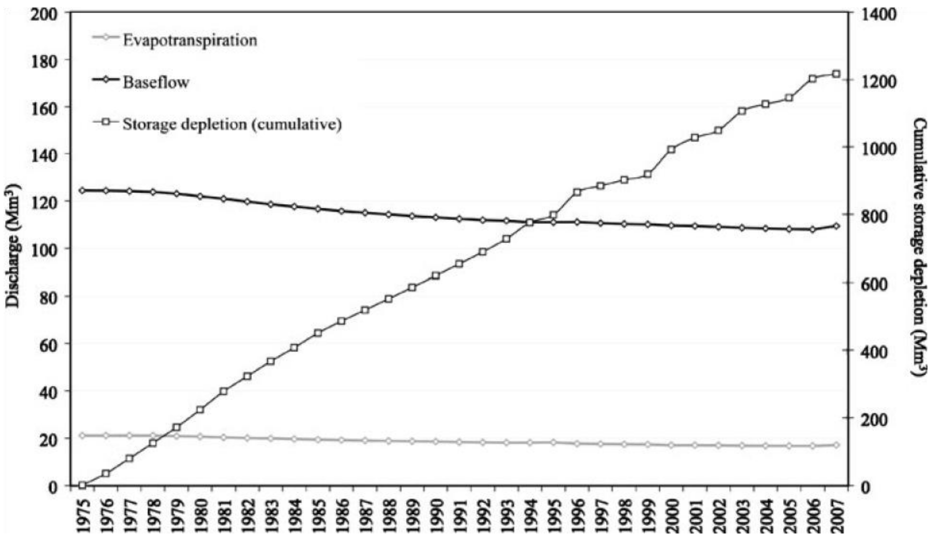


Fig. 8 Pumping patterns for forecast scenarios



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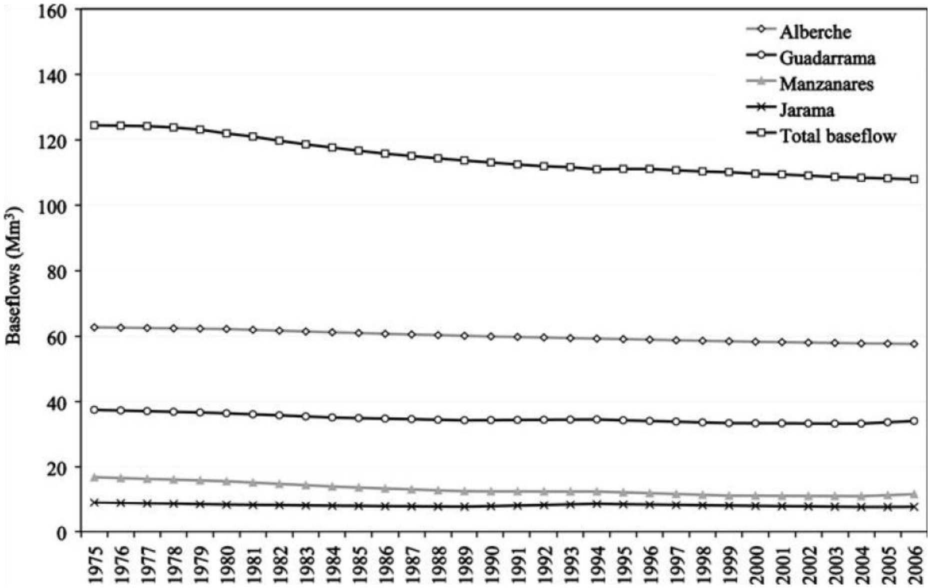

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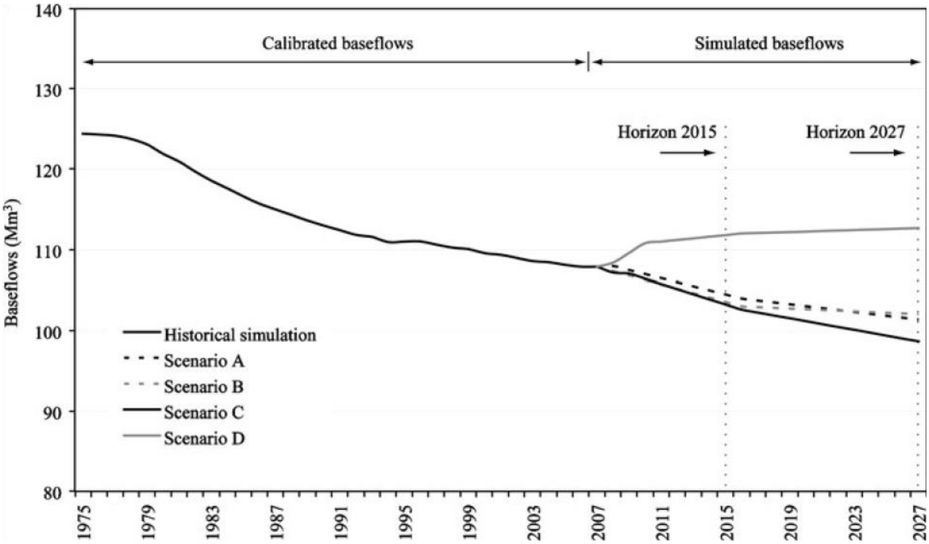

Fig. 11 Calibrated and simulated baseflows. Discharge is in expressed in yearly intervals

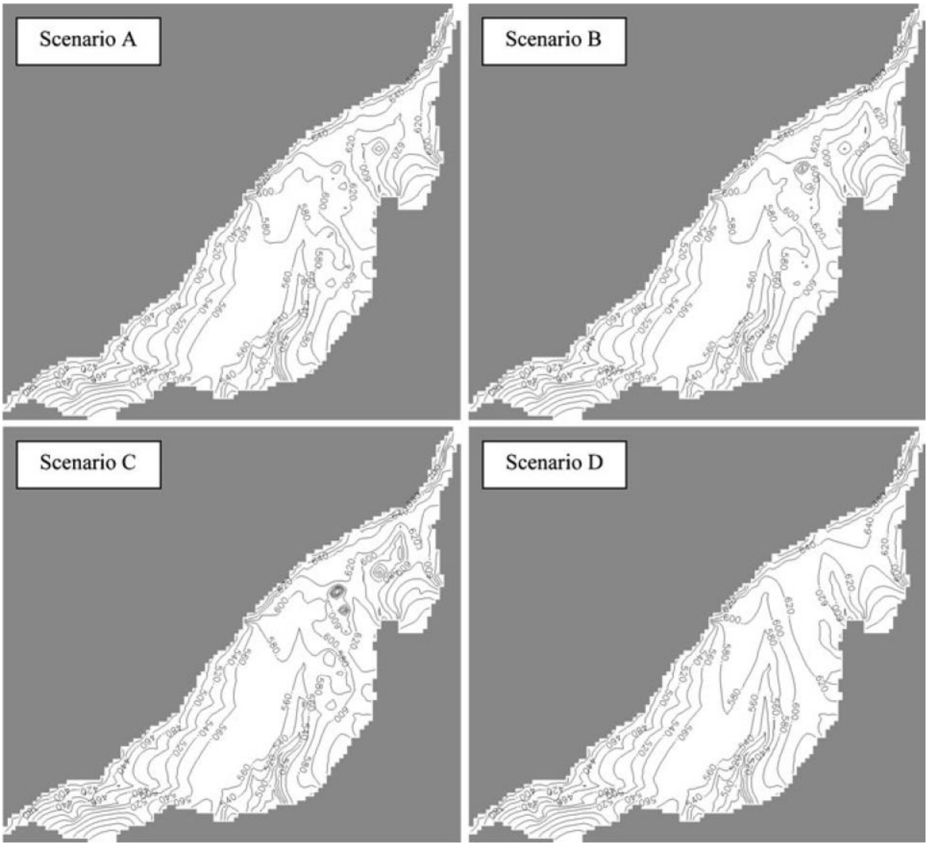

Fig. 8 Water table elevations by the end of each simulation period (2027)

Table 1 Water budget results for historical and forecast model runs

Water budget		Natural	Historical						Scenario		
			1976	1985	1995	2000	2005		2007	2015	2027
Outflows	Well	0.0	35.5	45.9	83.3	54.3	77.3	A	130.0	52.0	48.0
								B	150.0	150.0	150.0
								C	133.0	133.0	133.0
								D	0.0	0.0	0.0
	Discharge through streams	125.0	124.0	117.5	113.6	110.3	108.9	A	109.9	106.0	103.3
								B	109.4	104.9	103.8
								C	109.6	104.8	100.8
								D	112.0	113.5	114.2
	Evapo-transpiration	21.1	19.9	19.4	18.2	17.1	16.7	A	17.2	16.6	16.1
								B	17.1	16.4	16.0
								C	17.1	16.4	15.7
								D	17.4	17.7	17.8
Inflows	Recharge	145.5	144.5	144.5	144.5	144.5	144.5	A	144.5	144.5	144.5
								B	144.5	144.5	144.5
								C	144.5	144.5	144.5
								D	144.5	144.5	144.5
	Stream losses	0.1	0.7	0.8	0.4	0.7	0.8	A	0.9	1.1	1.1
								B	1.0	0.9	0.8
								C	1.0	1.2	1.5
								D	0.6	0.5	0.5

All figures are given in Mm^3/year

Table 2 Baseflow statistics per river sub-basin (Mm^3/year)

Parameter/stream	Alberche	Guadarrama	Manzanares	Jarama
Mean	60.0	34.7	13.1	8.0
Maximum	62.7	37.4	16.8	8.9
Minimum	57.6	33.2	11.0	7.5
Std. deviation	1.7	1.3	1.9	0.4
Variance	2.8	1.7	3.5	0.2
Current baseflow as % of natural baseflow	90.1	89.8	67.3	83.8
% baseflow decrease (1975–2006)	9.9	10.2	32.7	16.2
Baseflow as % of avg observed streamflows (1975–2007)[a]	10.7	19.5	2.8	1.2

Results for Alberche and Jarama respectively refer to the eastern and western margin of each stream
[a]Streamflows correspond to average yearly values and are measured at gauging stations in Escalona, Bargas, Rivas-Vaciamadrid and Mejorada del Campo

Table 3 Breakdown of forecast baseflows per subcatchment

Parameter/stream	Alberche	Guadarrama	Manzanares	Jarama
Natural baseflow (Mm^3/year)	62.7	37.4	16.8	8.9
Current baseflow as % of natural baseflow	90.1	89.8	67.3	83.8
Predicted baseflow as % of natural baseflow (2027—A)	90.0	80.6	54.5	69.5
Predicted baseflow as % of natural baseflow (2027—B)	90.0	83.2	51.2	74.3
Predicted baseflow as % of natural baseflow (2027—C)	90.0	78.8	44.5	64.7
Predicted baseflow as % of natural baseflow (2027—D)	90.3	94.3	79.6	94.1

Table 4 Comparative statistical overview of baseflows and streamflows per subcatchment (data for 1975–2006)

Parameter	Alberche		Guadarrama		Manzanares		Jarama	
	Stream flows[a]	Base flows	Stream flows[a]	Base flows	Stream flows[a]	Base flows	Stream flows[a]	Base flows
Mean	561.5	60.0	178.4	34.7	460.9	13.1	642.4	8.0
Maximum	1,407.5	62.7	524.9	37.4	845.6	16.8	1,677.0	8.9
Minimum	147.0	57.6	44.6	33.2	250.7	11.0	182.0	7.5
Std. dev.	398.6	1.7	159.6	1.3	131.9	1.9	436.5	0.4
Baseflow as % of avg streamflows	10.7		19.5		2.8		1.2	

Figures in Mm^3/year

[a] Streamflows correspond to average yearly values and are measured at gauging stations in Escalona, Bargas, Rivas-Vaciamadrid and Mejorada del Campo

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