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Modeling seawater intrusion in overexploited aquifers in the absence of sufficient data: application to the aquifer of Nea Moudania, northern Greece

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Abstract In many coastal areas, overexploitation of groundwater resources has led both to the quantitative degradation of local aquifers and the deterioration of groundwater quality due to seawater intrusion. To investigate the behavior of coastal aquifers under these conditions, numerical modeling is usually implemented; however, the proper implementation of numerical models requires a large amount of data, which are often not available due to the time-consuming and costly process of obtaining them. In the present study, the investigation of the behavior of coastal aquifers under the lack of adequate data is attempted by developing a methodological framework consisting of a series of numerical simulations: a steady-state, a false-transient and a transient simulation. The sequence and the connection between these simulations constitute the backbone of the whole procedure aimed at adjusting the various model parameters, as well as obtaining the initial conditions for the transient simulation. The validity of the proposed methodology is tested through evaluation of the model calibration procedure and the estimation of the simulation errors (mean error, mean absolute error, root mean square error, mean relative error) using the case of Nea Moudania basin, northern Greece. Furthermore, a sensitivity analysis is performed in order to minimize the error estimates and thus to maximize the reliability of the models. The results of the whole procedure affirm the proper implementation of the developed methodology under specific conditions and assumptions due to the lack of sufficient data, while they give a clear picture of the aquifer's quantitative and qualitative status.

Introduction

Coastal areas are often characterized as the most densely populated areas in the world (Post 2005; Sefelnasr and Sherif 2014) in the extent of which various human activities such as tourism development, commercial and agricultural activities, take place. In order to meet the ever-increasing water needs in these areas, high water demands are generated, thus causing the excessive exploitation of groundwater reserves and consequently the intense decline of groundwater levels (Datta et al. 2009). Under these stressful conditions, the natural balance existing between fresh and salt water is disturbed and, eventually, seawater intrusion occurs (Kopsiaftis et al. 2009; Narayan et al. 2007; Werner et al. 2013).

Seawater intrusion has evolved into a widespread environmental problem greatly affecting the groundwater potential of coastal areas (Lu et al. 2013; Werner and Gallagher 2006) and producing serious consequences for the environment, ecology and economy of these regions (Datta et al. 2009). For this reason, seawater intrusion constitutes the subject of thorough research for both water resource agencies and scientists; however, it is a complex hydrodynamic process, which depends on the effect of various factors (e.g. hydrodynamic dispersion and density variations, tidal effects, aquifer characteristics, recharge and discharge conditions) that render its study not a simple procedure (Bear 1999; Mao et al. 2006; Narayan et al. 2007; Rahmawati et al. 2013; Werner et al. 2013).

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The use of numerical modeling, which has evolved over the past 40 years into a powerful and efficient tool used widely in order to describe such a multifunctional phenomenon like seawater intrusion, plays a significant role in the proper understanding of the behavior of coastal aquifer systems (Cobaner et al. 2012; Dausman et al. 2010; Don et al. 2005). In general, two different modeling approaches have been developed in order to simulate seawater intrusion: the sharp-interface and the transition zone approximations (Bear and Cheng 2010; Reilly and Goodman 1985; Werner et al. 2013). Of these approximations, density-dependent modeling is far more complex but it better reflects reality (Lin et al. 2009; Rao et al. 2004; Van Camp et al. 2014), even though discrepancies may exist between density-dependent models and observed salinity profiles (Abarca et al. 2007; Sanford and Pope 2010).

With regard to this type of model, the improvement of knowledge concerning the various physical processes of seawater intrusion combined with the rapid evolution of computer science have led to the development of a large number of numerical codes. Most codes incorporate both flow equations, taking into consideration the effect of density variations, and solute mass transport equations, which are quite complex and require long computational time to be solved (Post 2005; Werner et al. 2013). SEAWAT (Guo and Langevin 2002; Langevin et al. 2003), SUTRA (Voss and Provost 2002) and FEFLOW (Diersch 2002) are among the most widely applied codes of this category.

Nowadays, numerical models constitute a powerful and valuable tool in the effort of comprehending seawater intrusion processes, studying the temporal and spatial evolution of seawater encroachment and rationally managing coastal aquifer systems. Nevertheless, various problems and difficulties persist to hinder the proper implementation of models and reduce their reliability and predictive capability. Data deficiency, as well as difficulties in defining the initial conditions, especially regarding the mass transport simulations, are included among the most common problems encountered in seawater intrusion research (Datta et al. 2009; Oude Essink 2003; Werner et al. 2013).

The proper and accurate application of numerical models quite often requires the existence of a plurality of different types of data with regard to the problem under study (flow and/or transport). In many cases, this is not feasible due to the time-consuming and costly process of obtaining them, as well as due to accessibility constraints (Narayan et al. 2007; Oude Essink 2003; Sanford and Pope 2010; Sherif et al. 2012). The problem becomes even more intense in seawater intrusion modeling, since the proper determination of the position and shape of the transition zone requires measurements of mass concentrations at different depths of the groundwater system, which in general are very difficult to undertake. Additionally, in order to better represent the temporal evolution of seawater intrusion, long time series of solute mass concentrations are required, leading to a costly procedure including a large number of measurements that have to be performed at regular intervals and cover a wide period of time (Oude Essink 2003). As Werner et al. (2013) report, the greatest shortfall in seawater intrusion research is the lack of monitoring studies in which the proper delineation and prediction of the transition zone changes are accomplished. Moreover, few models are calibrated and verified rigorously due to the limited field observations and the absence of adequate measurements with regard to solute concentrations.

Apart from data deficiency, defining the initial salinity field, i.e. determining the initial distribution of mass concentrations, is a serious problem for seawater intrusion modeling, since it is characterized as a difficult and not a straightforward procedure which is hindered due to the lack of data. Yet, it is of high importance since it greatly affects the model results, and therefore the accuracy of its predictions (Doherty 2008; Langevin and Zygnerski 2013; Sanford and Pope 2010; Werner et al. 2013). Various techniques have been developed and applied in order to deal with this kind of weakness of seawater intrusion modeling.

A common technique is to run a simulation until equilibrium is reached considering predevelopment (pre-pumping) conditions in a salt-free aquifer (false transient simulation) and then introduce the resulting salinity field as initial conditions to a subsequent transient simulation (Carrera et al. 2010; Dausman et al. 2010; Langevin and Zygnerski 2013; Sanford and Pope 2010). The main drawbacks of this technique are: (1) the requirement of many years to reach equilibrium, which corresponds to long time simulations and (2) the fact that usually the resulting salinity field does not match the measured concentrations in observation wells (Carrera et al. 2010; Sanford and Pope 2010; Zhang et al. 2004). An alternative approach involves the estimation of initial conditions as part of the model's calibration procedure, especially by the use of inverse modeling (Carrera et al. 2010; Iribar et al. 1997). Doherty (2008) outlines the basic principles of incorporating initial conditions into inverse modeling and denotes that two main problems have to be addressed in order to apply the methodology properly. The first is related to the parametric representation of the initial conditions, while the second refers to the adjustment of the large number of parameters that are introduced due to the aforementioned parametric representation (Doherty 2008), which in turn results in a marked increase of model complexity (Carrera et al. 2010).

A third technique attempts to create a realistic transition zone that also reflects the measured concentrations in the observation wells and is described thoroughly by Sanford et al. (2009) and Sanford and Pope (2010). According to this method, even though the simulation is run to equilibrium, snapshots of the chloride-concentration field are saved at various time steps based on the comparison between simulated and observed values. In this way, a snapshot of a chloride-

concentration field for the entire model area is associated with each well. These multiple fields are then combined into a smoothly varying field by using an interpolation method (e.g. Kriging technique).

The present study investigates the operation of coastal aquifer systems characterized by overexploitation conditions, and therefore by seawater intrusion through the application of numerical modeling and in the light of absence of sufficient data. This absence of data could be related both to the conceptual model of the aquifer under consideration and the variables of the problem under study. The methodological framework which is presented consists of the development of a series of numerical simulations by using the widely applied codes MODFLOW (Harbaugh et al. 2000; McDonald and Harbaugh 1988), MT3DMS (Zheng and Wang 1999) and SEAWAT (Guo and Langevin 2002; Langevin et al. 2003). In addition, as an important part of the whole methodological procedure, the determination of the initial conditions for the transient simulation, especially regarding the transport problem, is attempted. To this task, the technique described by Sanford and Pope (2010) is applied under specific modifications due to the lack of sufficient data, while maintaining its efficiency and its main advantages (i.e. creating a realistic salinity field which reflects the measurements in observation wells).

The proposed methodology constitutes a generic, yet simple enough and quite efficient, methodology in the effort of dealing with seawater intrusion problems where no adequate data are available. Its successful implementation directly results in a satisfactory simulation of the physical processes of groundwater flow and seawater intrusion, as well as a sufficient projection in time of both hydraulic head and chloride concentrations distributions.

The pilot implementation of the proposed methodology is performed in the aquifer of Nea Moudania in the Halkidiki Peninsula (northern Greece), which is marked by severe quantitative and qualitative problems that are directly linked to the intensification of agricultural activities in the region. Such activities produce an excessive exploitation of groundwater resources, which in turn results in both increased drawdown and intense seawater intrusion. Simulating groundwater flow and seawater intrusion in the aquifer of Nea Moudania assists in better understanding its dynamics and in creating a useful tool for the proper planning and implementation of management policies.

Methodology

The proposed methodology aims at investigating the behavior of coastal aquifer systems under the absence of sufficient data by conducting a series of numerical simulations—more specifically, a steady state, a false transient and a transient simulation are carried out. Each of these simulations has its own specific objectives, while the connection between the false transient and the transient simulation leads to the determination of the initial salinity field for the latter one in a way which constitutes a simplification of the technique described by Sanford and Pope (2010).

The implementation steps of the proposed methodology are the following: (1) development of models based on the available data regarding the geological, hydrogeological and hydrological conditions of the area under study, taking into consideration the type of the simulation (i.e. steady-state or transient), (2) calibration of models with the limited available data by making the proper adjustments on the aquifer parameters (particularly on those of high uncertainty and of significant lack of data and measurements) and applying a repetitive trial-and-error process between the false transient and the transient simulation and (3) sensitivity analysis, which is performed in order to investigate the influence of the models' input parameters to the results of the calibration process.

Since the proposed methodology is based on developing a series of numerical simulations, it is readily understood that its evaluation and its validity check focus essentially on evaluating the reliability of the aforementioned simulations, which in turn are inextricably linked to the evaluation of their calibration process—a procedure which, in the context of this study, is accomplished by estimating specific accuracy measures, including the mean error (ME), the mean absolute error (MAE), the root mean square error (RMSE) and the mean relative error (MRE), which provide the error size of each simulation directly and effectively, and hence the degree of its reliability. Therefore, as long as these errors are minimized, the reliability of each simulation is increased and the robustness of the proposed methodology is enhanced.

At this point, it is important to mention that all the aforementioned simulations are performed in a two-dimensional (2D) horizontal plane mainly due to the fact that all available data, and especially the solute-concentration measurements, refer to mixed water abstracted from the whole thickness of the aquifer (Iribar et al. 1997; Pool et al. 2015). Furthermore, the scope of this work is not to define the exact shape and location of the transition zone, but firstly to investigate its extension and movement and then to designate the portion of the aquifer affected by seawater intrusion over time in order to create a useful management tool (Sefelnasr and Sherif 2014; Werner and Gallagher 2006).

Model development procedure

Steady-state simulation

The steady-state simulation constitutes the primary simulation of the whole model development procedure and includes the simulation of groundwater flow under steady-state conditions

by applying the MODFLOW code. It aims at forming a generic image with regard to the behavior of the aquifer under study, as well as at achieving an initial estimation concerning specific aquifer parameters (e.g. hydraulic conductivity, recharge). Since the simulation is performed under steady-state conditions, no temporal discretization is required, while boundary conditions remain constant and recharge and discharge components are calculated at an annual base.

False transient simulation

The false transient simulation follows the steady-state one and includes the simulation of both groundwater flow and solute transport under transient conditions by applying the SEAWAT code and taking into consideration the density variations due to the variations in the solute concentrations. The simulation is run until it reaches equilibrium with average aquifer stresses (i.e. recharge, pumping), constant boundary conditions and negligible storativity in order to obtain steady-state conditions (Ganesan and Thayumanavan 2009; Rao et al. 2004; Zimmermann et al. 2006). The simulation time step is selected so that the computational time will not be too long, as well as the difference in concentration values between two consecutive time steps will be within certain limits. Finally, freshwater conditions are assumed everywhere as initial conditions for the transport problem, since the intrusion of seawater wedge commences at the beginning of the simulation. This simulation is performed because the SEAWAT code cannot be applied under steady-state flow conditions and in order to get the initial conditions for the transient simulation which follows.

Transient simulation

The transient simulation constitutes the main simulation of the model development procedure and involves the simulation of both groundwater flow and solute transport under transient conditions by applying the SEAWAT code. This simulation is considered more realistic, since aquifers under overexploitation conditions are not in dynamic equilibrium (Ahmed and Umar 2009; Oude Essink 2003). Its main purpose is to investigate the aquifer response during different time periods under specific recharge and discharge conditions, as well as to predict the distribution of both hydraulic head and solute concentrations over time, which would lead to the rational and proper management of the aquifer. Since the simulation is performed under transient conditions, temporal discretization is required, while when necessary, the input parameters (e.g. recharge and discharge components, boundary conditions) are introduced into the model based on the aforementioned temporal discretization. Finally, the initial conditions of the transient simulation are obtained from the previous false transient simulation through a procedure which is described thoroughly in the following.

Calibration procedure

The calibration procedure includes two main steps. At first, the steady-state model is calibrated independently from the other two models and a primary adjustment of specific aquifer parameters (i.e. hydraulic conductivity, recharge) is made. To this task, the automated inverse model calibration tool PEST (Doherty 2002) is applied. Then, the calibration of the other two models is performed through a repetitive trial-and-error procedure in order to get the minimal simulation errors regarding both the flow and the transport problem. This is the most complex and time-consuming step, yet at the same time, the most important one of the methodology described in this section. It is aimed at determining the initial conditions regarding the transient simulation, as well as at properly adjusting the various aquifer parameters—i.e. hydraulic conductivity, storativity, effective porosity, dispersivity, recharge, pumping.

More specifically, the aforementioned calibration procedure includes the following steps (Fig. 1): (1) the adjustment of the aquifer parameters associated with groundwater flow (e.g. hydraulic conductivity) through the calibration process of the false transient simulation, (2) the adjustment of the aquifer parameters associated with solute mass transport (e.g. effective porosity, dispersivity) through the calibration process of the false transient simulation, (3) the calibration of the false transient model for a specific time step (before equilibrium is reached), during which a satisfactory correspondence between model results (hydraulic head, solute concentrations) and observed field data is obtained (both hydraulic head and concentration distribution match the observed values for this specific time step), (4) the introduction of the false transient simulation results (hydraulic head, solute concentrations) corresponding to the selected time step into the transient simulation as initial conditions, (5) the adjustment of the aquifer parameters associated with groundwater flow (e.g. storativity, recharge, pumping) through the calibration process of the transient simulation, and (6) the testing of the transport parameter values estimated in the preceding (step 2), through the calibration procedure of the transient simulation.

In the latter case, the testing aims to examine whether the results of the transient simulation are consistent with the measured values in the observations wells as far as the values of the transport parameters estimated in the false transient simulation are concerned. If this occurs, then the transient model can be used for projection. Otherwise, the values of transport parameters are modified in the false transient simulation, which is set anew and recalibrated and therefore a new, different time step is chosen, the results of which are introduced as initial conditions into the transient simulation. This process is repeated until the calibration procedure provides satisfactory results (minimal simulation errors).

What is described in the preceding constitutes the main difference between the technique presented by Sanford and
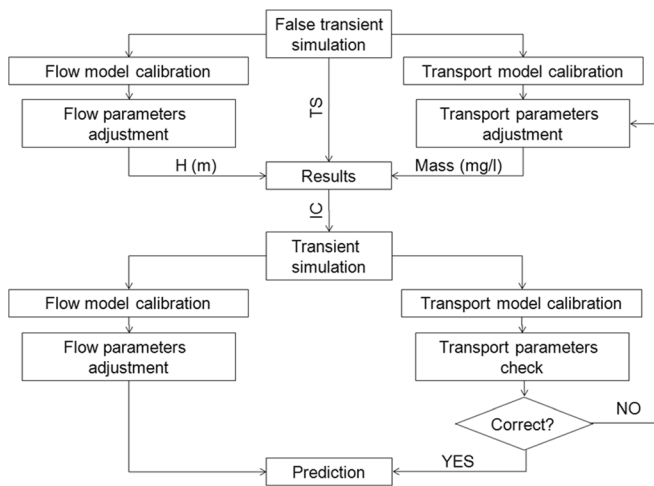
Fig. 1 Schematic representation of the calibration procedure of the false transient and the transient simulations (*TS* time step, *IC* initial conditions, *H* hydraulic head)

Pope (2010) and the proposed methodology regarding the determination of the initial salinity field. More specifically, while the technique described by Sanford and Pope (2010) uses snapshots of the solute-concentration field saved at various time steps, the proposed methodology uses a specific time step of the false transient simulation, the results of which are introduced into the transient simulation. This renders the proposed methodology simpler, but still efficient, since it attempts to create a realistic salinity field based on the available data. At this point, it is worth mentioning that the limited availability of measurements regarding the solute concentrations has triggered the development and the implementation of the proposed methodology, which substantially attempts to compensate for the lack of sufficient data turning this lack into a prerequisite for its proper application.

Sensitivity analysis

The final step of the proposed methodology includes a sensitivity analysis in order to investigate the influence of the model's input parameters on the results of the calibration procedure of the main simulation, i.e. the transient simulation, and therefore to examine the reliability of the technique already described. To this task, the variation of the RMSE of the transient simulation was observed due to the variation of several model parameters (i.e. hydraulic conductivity, storativity, effective porosity, recharge, pumping). This procedure was implemented by modifying only one input parameter at a time while keeping the others constant (Don et al. 2005). Through this process, better adjustment of the aforementioned parameters is achieved, since in the case of an error increase, which occurs by modifying the value of a certain parameter, this parameter is redefined so that the error decreases. This procedure empowers the proposed methodology and strengthens its implementation, while it allows for studying the influence of model parameters

on model results, as well as for gaining insight into model behavior (Carrera et al. 2010; Rajabi et al. 2015).

Study area

The hydrological basin of Nea Moudania extends to the south-western part of the Halkidiki Peninsula (south-east of the city of Thessaloniki). It is part of a larger region known as "Kalamaria Plain", constituting the prime agricultural area of Halkidiki, which has been intensively cultivated and irrigated, thus generating increased water demands. The catchment covers about 127 km^2, with a mean soil elevation of 211 m (above sea level) and a mean soil slope of 1.8 %. It actually constitutes a coastal hydrological basin which is in direct hydraulic connection with the sea (Latinopoulos 2003; Siarkos and Latinopoulos 2012; Xefteris et al. 2004).

Figure 2 depicts both the location and the boundaries of the Nea Moudania basin, which is divided into two sub-regions, the hilly area in the north (46.3 %) and the flat area in the south (53.7 %). The climate of the study area is semi-arid to humid, typically Mediterranean, and the average annual precipitation is 417 mm for the flat area and 504 mm for the hilly one. It is characterized by a scalable elevation of the terrain from the coastal to the inland area and a dense hydrographic network, especially in the hilly area, draining directly to the sea. The streams in this network are of occasional flow resulting from the surface runoff that discharges into the sea. Almost 76 % of the study area is used as agricultural land, 20 % is woodland (mostly the northern part) and the remaining 4 % accounts for urban and touristic development (Latinopoulos 2003; Siarkos 2015).

With regard to the geology of the study area, the larger part of the region is located within the Peonia geologic zone and more specifically within the Moudania geologic formation. This formation constitutes part of the Neogene sediments located in Western Halkidiki, which are developed mostly in low relief sections covering pre-Neogene background. These Neogene sediments are composed mainly of alternated beds of sandstones, conglomerates, sands, silts, and red to brick red clays. In addition, a small portion of the basin in the north-east lies within the Circum-Rhodope Belt where rocky formations (e.g. schists, gneisses, ophiolites) are located. Finally, along the largest streams of the region and mainly at their estuaries (i.e. south part of the basin), Quaternary deposits consisting of sands, grits and clays are developed (Latinopoulos 2003; Syridis 1990; Xefteris et al. 2004).

Based on the preceding analysis, the geological formations of the study area are classified into two main groups regarding their hydrological behavior: (1) the loose and (2) the rocky formations. Of these categories, loose formations, and especially the Neogene sediments (Moudania formation), are of greater hydrological interest due to their great extension, their
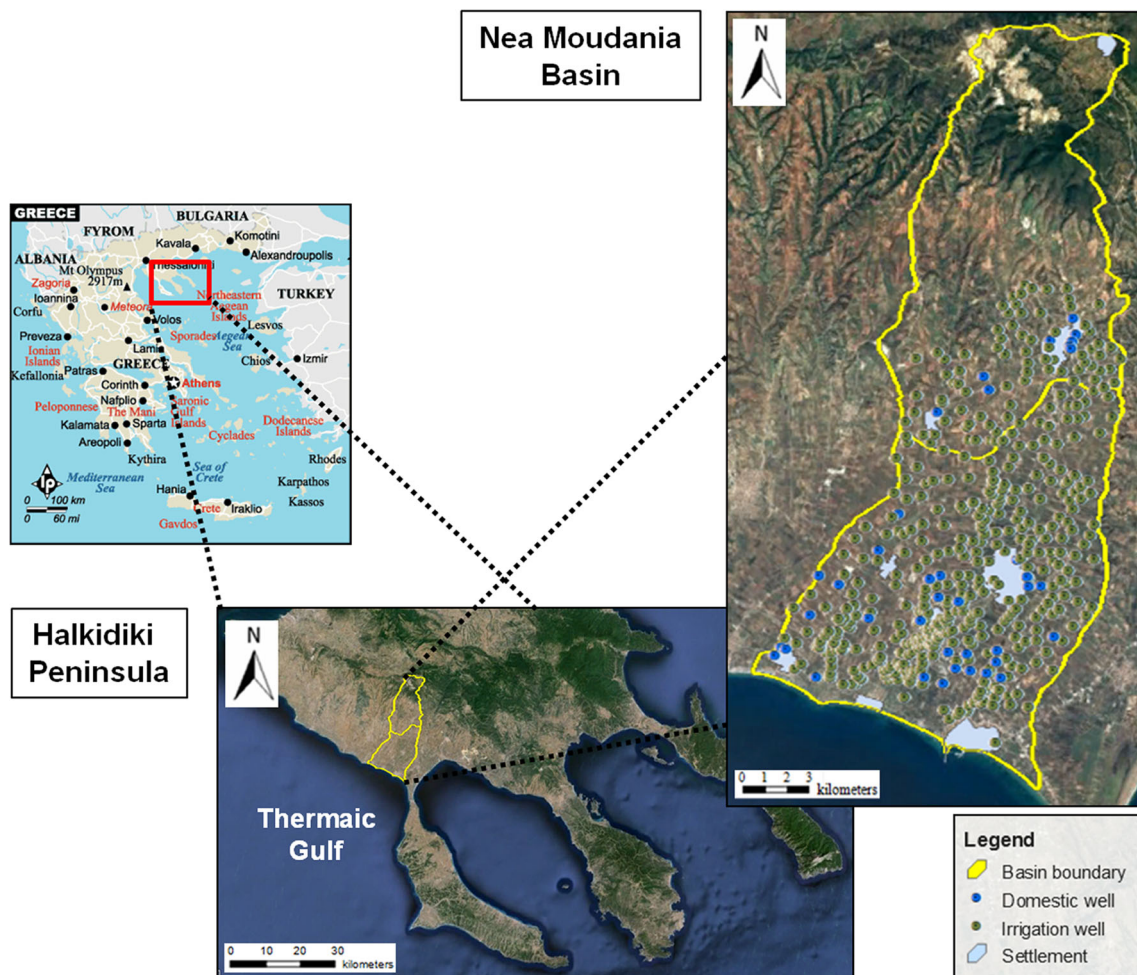
Fig. 2 Location and boundaries of Nea Moudania basin

considerable thickness and their high water-storage capacity. Apart from the fact that Moudania formation occupies most of the study area, it constitutes the main aquifer, which is exclusively used and therefore intensively exploited. Under natural conditions, the Nea Moudania aquifer is a semi-confined aquifer system which consists of successive water-bearing layers separated by lenses of semi-permeable or impermeable materials (Fig. 3). Nevertheless, the overexploitation conditions existing in the whole region have caused substantial decline of the groundwater levels, thus creating phreatic conditions in several locations.

The whole basin is a typical rural area, where agriculture dominates both the local economy and land use. As a result, the water used for irrigation purposes sums up to approximately 94 % of the total water consumption in the region; the remaining accounts for domestic (5.5 %) and livestock (0.5 %) use. All these water needs are solely satisfied by the aforementioned aquifer system, and therefore a large amount of private and municipal wells operate in the region. Referring to the year 2001, there are totally 518 wells (Fig. 2; about four wells per km², 479 of which are private irrigation wells

clustered mainly in the central part of the flat area. Irrigation well pumping is arbitrary and uncontrollable leading to continuous and intense exploitation of the local groundwater resources and considerably modifying the groundwater regime in the region. The remaining wells are water supply wells which are also used to serve livestock needs (Latinopoulos 2003; Latinopoulos and Siarkos 2014).

In general, the study area is characterized by a significant quantitative degradation of its groundwater resources and a net deficit in the aquifer water balance, since the total water demand exceeds natural recharge (Latinopoulos 2003). Besides this fact, the qualitative degradation of groundwater is considered to be an issue of high importance, as it causes substantial reduction in the usable quantities of water and threatens the health of the local population (Siarkos et al. 2014). Seawater intrusion is one of the main causes contributing to the deterioration of the quality of the local groundwater resources. This is a significant problem along the coastline, which escalates with time due to the overexploitation conditions prevailing in the whole region. According to Latinopoulos (2003), large cones of depression are formed
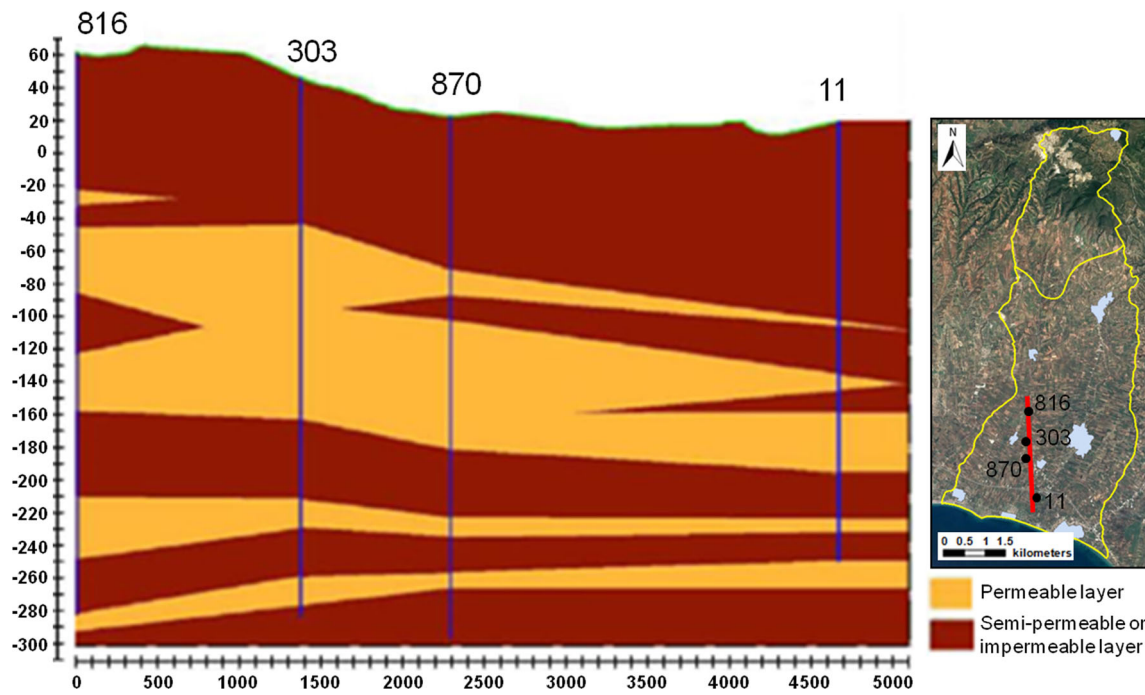
Fig. 3 Schematic geological section across the study area (adapted from Latinopoulos 2003)

in the central part of the flat area where groundwater levels fall below the mean sea level due to the intense pumping for irrigation purposes, leading to the reversal of the natural groundwater flow and the inflow of seawater along the coastline and through the interior of the region.

Conceptual model

In the following sections, a detailed description of the main components of the conceptual model of Nea Moudania aquifer is provided, taking into account both the simulation of groundwater flow and seawater intrusion. To this task, information is provided regarding the geometry and the boundary conditions of the aquifer system, its hydraulic-hydrodynamic parameters (i.e. hydraulic conductivity, storativity, effective porosity, dispersivity), as well as its recharge and discharge conditions (prior estimates). All the necessary data are derived from previous research conducted in the area (Latinopoulos 2003), while the whole conceptualization procedure constitutes an improvement of the ones described in Siarkos and Latinopoulos (2012) and Latinopoulos and Siarkos (2014).

Aquifer geometry

The boundaries of the aquifer under study match the boundaries of the Nea Moudania basin, with the exception of the north-eastern part, which consists mainly of rock formations, rendering the division of the aquifer and the use of a portion of it during the development of the various models

necessary (Fig. 4). The determination of the dividing boundary is not related to the physical boundaries of the aquifer, but it relies entirely on the piezometric conditions in the study area. Therefore, the dividing boundary corresponds to the piezometric contour of 150 m referring to the year 2001, which was obtained applying the Kriging method (Latinopoulos 2003).

Regarding the aquifer thickness, it is assumed that the various successive permeable layers form a single, unified aquifer system, since all the simulations are performed in a 2D horizontal plane. Based on information derived from various well logs and geological sections, a uniform thickness of 250 m was assumed. The upper limit of the aquifer was set at the depth at which the semi-permeable coating reaches below the ground level (an average value of 50 m was obtained based on well logs), while the lower one resulted from the upper by subtracting the assumed uniform thickness of the aquifer.

Boundary conditions

The determination of the aquifer boundary conditions, as far as the flow problem is concerned, was based on data referring to the hydrogeology and the piezometric conditions in the study area (Fig. 4). The eastern and western boundaries were assigned as no flow boundaries, since there is no hydraulic connection between the aquifer under study and the neighboring regions and the flow lines are parallel to these boundaries according to the general groundwater flow direction on a regional scale (Latinopoulos 2003). The southern
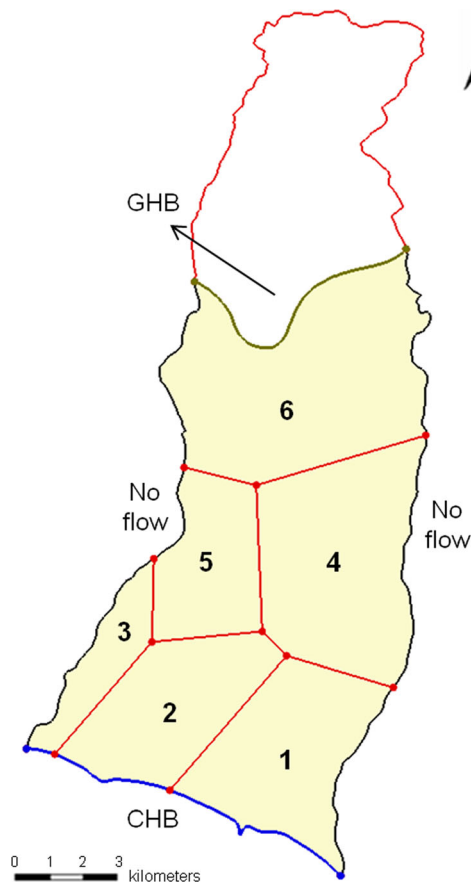
Fig. 4 Flow boundary conditions (*CHB* constant head boundary: *blue line*, *GHB* general head boundary: *green line*, *No flow* no flow boundaries: *black line*) and the six distinct zones of the study area

Hydraulic-hydrodynamic parameters

The determination of the aquifer hydraulic-hydrodynamic parameters such as hydraulic conductivity, storativity, effective porosity and dispersivity, was based on data deriving both from previous research conducted in the study area (Latinopoulos 2003) and from the respective literature, taking into account the type of geological formations existing in the region. Data from previous research include the results of a few pumping tests in individual wells that refer exclusively to values of hydraulic conductivity and storativity. The study area was divided into six distinct zones (based on the number of the tested wells) using the Thiessen Polygon method and constructing polygons around the data points (Fig. 4). Each of these zones was assigned a different parameter value based on the pumping tests results.

According to the pumping tests, hydraulic conductivity ranges between 0.125 and 0.240 m/day, while storativity ranges between 0.03 and 0.10. However, hydraulic conductivity values vary from 0.086 to 1.730 m/day as far as the broader research area is concerned (Latinopoulos 2003), while with respect to storativity no other information exists. Therefore, the value range of this parameter was based not only on pumping tests but also on literature for similar types of sediments (Freeze and Cherry 1979; Soulios 1996) and was set between 0.01 and 0.15.

For the transport parameters, i.e. effective porosity, dispersivity (both longitudinal and transverse) and molecular diffusion, there are no available data and therefore it was essential to quantify them in another way. The definition of effective porosity was based exclusively on specific yield, and therefore a value range between 0.05 and 0.15 was considered. The relationship for longitudinal dispersivity $\alpha_L = 0.83 \times [\log(L)]^{2.414}$ (Almasri and Kaluarachchi 2007), where L is the spatial scale of the groundwater flow (m), was used in order to calculate longitudinal dispersivity, resulting in a value of 60 m, which was assigned to the whole region. The ratio of transverse to longitudinal dispersivity was taken as 0.1 (Cobaner et al. 2012; Giambastiani et al. 2007; Kerrou et al. 2010), while molecular diffusion was considered to be negligible.

Finally, it should be noted that all the aforementioned parameters, and especially those with no available measurements, are characterized by a high degree of uncertainty. This renders their adjustment as part of the calibration process of the various simulations necessary, by using the measured, estimated or assigned values as an initial guess and taking into consideration their value range according to both the available data and the literature. All the parameters were calibrated separately for each individual zone (Fig. 4), allowing a better fit of available observations (Kerrou et al. 2010), with the exception of dispersivities, which were assumed to be homogenous along the whole aquifer.

boundary was simulated as a constant head boundary (CHB), where the hydraulic head was set to zero, since in this section the aquifer is in direct hydraulic connection with the sea. The northern boundary was delineated as a general head boundary (GHB) in order to prevent unrealistic effects on simulated heads (Langevin 2003; Sanford and Pope 2010). The hydraulic head along this boundary was set equal to 150 m, based on the piezometric conditions in the study area regarding the year 2001. Moreover, regarding this type of boundary (GHB), conductance was defined through the calibration process of the steady-state simulation. Finally, no flow boundary conditions were assigned to the bottom of the aquifer system.

With regard to the transport problem, the southern boundary is considered of high importance since it is in hydraulic connection with the sea. This boundary was simulated as a constant concentration boundary (CCB), where concentration was set equal to 19,000 mg/L, since chloride ions are used as a tracer for the characterization of groundwater salinization. With respect to the remaining boundaries either no water inflow is observed and no dissolved mass as well (eastern and western boundaries) or the chloride concentrations are considered particularly low (northern boundary).

Recharge conditions

The Nea Moudania aquifer is naturally recharged mainly by rainwater and, to a lesser degree, by irrigation return flows and losses from both water supply and wastewater networks or septic systems. Moreover, what is worth mentioning is the fact that the aquifer is also recharged both from the southern and northern boundaries due to the hydraulic connection with the sea and the Neogene sediments, respectively. The test of this consideration, as well as the estimation of the exact amount of water deriving from the aforementioned boundaries are made through the simulation of groundwater flow and more specifically through the determination of the aquifer flow budget.

Concerning the aquifer recharge from rainfall, after subtracting the losses from evaporation and direct runoff, the amount of water infiltrating into the ground was estimated and introduced into the model with separate values, depending on which sub-region (flat or hilly area) it is related to. Moreover, two types of land use were taken into consideration, especially the agricultural and the urban ones, since woodland is observed mainly in the north part of the basin. Due to the high uncertainty of the aforementioned determination procedure, recharge from rainfall was subjected to calibration in the transient simulation. With respect to the other two parameters (i.e. irrigation return flows and water supply/wastewater network leakage), it was assumed that 15 % of irrigation water and approximately 50 % of the water consumed for domestic use returns to the aquifer.

Discharge conditions

The groundwater resources of the study area are exploited for irrigation, domestic and livestock purposes. Since the accurate calculation of water demand (i.e. the quantity of water potentially abstracted from the aquifer) presents challenges due to the absence of water meters, the annual consumption of water was estimated for each district in the region and for each use. Next, each quantity was divided by the number of operating wells for that specific use in order to estimate the average pumping rate per well; however, it is likely that actual pumping rates, especially in the case of irrigation wells, exceed the estimated values and therefore they were subjected to calibration in the transient simulation.

Numerical modeling

Spatial discretization

In the present study, the model grid was formed so that it fully corresponds both to the simulation conditions and to the assumptions made when developing the aquifer conceptual model and it was kept intact through the various simulations, since

they are linked together. More specifically, a regularly spaced, single layer model grid was constructed with equal-sized cells in the horizontal plane in order to minimize the effects of numerical problems occurring in solute transport models (Langevin 2003). Its construction was based entirely on the transport problem and therefore the Peclet number (Pe) was considered as the main criterion in order to avoid numerical dispersion and oscillation errors (Oude Essink 2003; Voss and Souza 1987; Werner and Gallagher 2006; Zhang et al. 2004).

As far as the finite-difference method is concerned, the following condition should be satisfied: $Pe \leq 2$ (Oude Essink 2003; Rao et al. 2004). In order to meet this condition and since the longitudinal dispersivity was estimated equal to 60 m (see section 'Hydraulic-hydrodynamic parameters'), the model grid was formed so that each cell has a 100-m side. This, in conjunction with the extent of the study area, leads to a grid that consisted of 180 rows and 120 columns, and a total number of 9,599 active cells.

Temporal discretization

Temporal discretization refers to the assignment of both the time duration and the time step of the false transient and transient simulations. The false transient simulation was performed with a time step of 50 days and was run until equilibrium was reached (after several decades). For the transient model a 33-year simulation period was selected (2001–2034), which was divided into 396 monthly stress periods. It also should be mentioned that pumping/irrigation (1st May–30th September) and non-pumping (1st October–30th April) periods were considered, since both recharge (irrigation return flows) and discharge (operation of irrigation wells) receive different values during each period.

Calibration procedure

Steady-state simulation

The model calibration was performed initially using the trial-and-error method in order to achieve an initial estimation regarding the adjusted parameters. Then, the automated inverse model calibration tool PEST was applied in order to optimize the parameter values calculated previously and, thus, to minimize the simulation errors. To this task, 17 observation wells monitored during November 2001 were used. The whole procedure went on until a best fit was obtained between the simulated and measured water levels.

False transient simulation–transient simulation

Through the procedure described in the following, the false transient and the transient models are linked together in order to properly obtain the initial conditions (hydraulic head,

chloride concentrations) for the transient model, as well as to achieve a satisfactory adjustment of the various model parameters (i.e. hydraulic conductivity, storativity, effective porosity, dispersivities, recharge, pumping rates). The procedure actually includes the calibration of the aforementioned models to a feasible extent based on the available data and consists of the following steps:

- Readjustment of hydraulic conductivity through the calibration of the false transient model on the basis of hydraulic head data deriving from the same 17 observation wells used for the calibration of the steady-state model. To this task, the trial-and-error methodology was used so that the simulated water levels match the ones measured in the field.
- Adjustment of effective porosity and longitudinal dispersivity through the calibration of the false transient model on the basis of chloride concentration data referring to the year 2001 and particularly to the fall season (September–November). As only one measurement is available from a well close to the coast, where increased chloride concentrations due to seawater intrusion (Cl = 983 mg/L) are observed, the determination of the salinity field was based on that value.
- Selection of a time step in the false transient simulation, the results of which are consistent with the aforementioned measurements with regard both to the hydraulic head and chloride concentrations. These results are introduced as initial conditions in the transient simulation.
- Adjustment of the remaining parameters regarding the flow problem (i.e. storativity, recharge, pumping rates, boundary conditions) through the calibration/validation of the transient model applying the trial-and-error methodology until a best fit is obtained between the simulated and observed water levels. To this task, observation wells monitored during November 2002 (13 wells; Latinopoulos 2003), April 2003 (12 wells; Latinopoulos 2003) and November 2010 (12 wells; Siarkos 2015) were used. The first two periods were used for the calibration of the model, while the last one for its validation.
- Testing the effective porosity and longitudinal dispersivity values resulting from the false transient model, through the calibration of the transient model and on the basis of chloride concentration measurements derived from observation wells monitored during April 2011 (1 well; Siarkos 2015), November 2011 (1 well; Siarkos 2015) and April 2014 (4 wells; Siarkos 2015). All these periods comprise the model calibration period regarding the transport problem due to the limited amount of data. The main purpose of this step is to examine whether the results of the transient simulation are consistent with the measured values in the observation wells as far as the values of the transport parameters estimated in the false transient simulation are concerned.

Several model runs were performed. In each case a different time step was selected until the minimum discrepancies between the calculated and observed chloride concentrations were observed.

At this point, it should be mentioned that the calibration of both models is not an autonomous procedure but it is dependent on the calibration of each one, since the parameter adjustment taking place in one model may have as a result the requirement for proper modifications in the other one. Therefore, the calibration process of the two transient models is in a dynamic state, where each model depends straightforwardly on the other. Moreover, even though several model runs and iterations were made, the computational time of the whole procedure is relatively short due to the lack of data and the simplicity of the various models.

Results

In this section the results of the models calibration are presented, as well as those of the sensitivity analysis, conducted in order to investigate the influence of the various parameters to the RMSE estimates of the transient model. Then, the results of the transient simulation, regarding both the flow and the transport problem, are displayed.

Calibration procedure

Steady-state simulation

The accuracy of the simulation was tested by calculating the mean error (ME), mean absolute error (MAE), root mean square error (RMSE) and mean relative error (MRE). The ME was found equal to 0.096 m, indicating that, on average, the simulated groundwater levels were slightly higher than the observed groundwater levels. The MAE, RMSE and MRE estimates were equal to 1.193 m, 1.419 m and 1.01 % respectively, signifying a rather successful calibration and therefore a satisfactory simulation. Moreover, through the calibration of the steady-state model, both hydraulic conductivity and conductance were adjusted and determined. More specifically, hydraulic conductivity in the six distinct zones of the study area (Fig. 4) ranged between 0.116 and 0.669 m/day (Table 1; maximum values occurred in coastal zones, i.e. zones 1, 2 and 3), while conductance was found equal to 65 m²/day (which was used in the other models as well).

False transient simulation–transient simulation

Similar to the previous case, the same statistical errors were used in order to check the validity of the false transient and the transient simulations regarding both the flow

and the transport problem. With respect to the false transient model and the flow problem, in the simulation that was considered calibrated, the ME, MAE, RMSE and MRE estimates were equal to −0.026 m, 1.151 m, 1.438 m and 1.02 % respectively, indicating a rather successful calibration and therefore a satisfactory simulation. As far as the transport problem is concerned, since only one measurement is available for the determination of the salinity field, there is no point in estimating the simulation errors. With regard to the transient model and the flow problem, in the simulation that was considered both calibrated and validated, the ME, MAE, RMSE and MRE estimates were equal to −0.176 m, 1.502 m, 1.735 m and 1.02 % respectively regarding the calibration period and −0.478 m, 1.822 m, 2.059 m and 1.36 % respectively regarding the validation period, indicating in general a rather satisfactory simulation. Concerning the transport problem, the ME, MAE, RMSE and MRE estimates were equal to −0.10 mg/L, 6.93 mg/L, 9.29 mg/L and 3.36 % respectively, indicating a quite satisfactory simulation.

Finally, through the calibration procedure of the two transient models, adjustment of the following parameters was made: (1) hydraulic conductivity, (2) storativity (both specific storage and specific yield), (3) effective porosity, (4) longitudinal dispersivity, (5) aquifer recharge from rainfall, (6) pumping rates of irrigation wells and (7) hydraulic head values in the north boundary. Regarding the aquifer hydraulic-hydrodynamic parameters, the results of their adjustment are presented in Table 1 for each of the six distinct zones of the aquifer under study (except for longitudinal dispersivity, which is considered homogenous through the whole aquifer system). Concerning the aquifer recharge from rainfall, the whole procedure resulted in a 15 % reduction compared to its initial estimates, which is considered reasonable due to the uncertainty of its estimation procedure. With respect to the pumping rates of irrigation wells, an increase of 17 % was made, which is also reasonable considering that the initially estimated values are characterized as rather conservative and are lower than the actual ones. In Fig. 5, the hydraulic head values at both ends of the aquifer's north boundary during the simulation period are presented.

Sensitivity analysis

The sensitivity analysis was carried out in order to test the validity of the calibration process by investigating the influence of the various model parameters to the RMSE estimates. Through this procedure and by minimizing the RMSE estimates, all the model parameters are tested, while their optimum adjustment is accomplished. The error estimates displayed previously are the final outcome of the calibration procedure after implementing the sensitivity analysis.

Table 1 Estimated values of the various aquifer parameters

Zone	Steady-state model Hydraulic conductivity (m/day)	Transient model			
		Hydraulic conductivity (m/day)	Effective porosity/ specific yield	Specific storage (m^{-1})	Longitudinal dispersivity (m)
1	0.647	0.575	0.150	0.00004	75
2	0.669	0.514	0.125	0.00012	
3	0.325	0.325	0.145	0.00019	
4	0.148	0.154	0.070	0.00015	
5	0.116	0.114	0.050	0.00006	
6	0.338	0.344	0.140	0.00020	

Figures 6, 7 and 8 present the influence of the model parameters, i.e. hydraulic conductivity (K), specific storage (S_s), specific yield/effective porosity (S_y/n_e), longitudinal dispersivity (α_L), aquifer recharge due to water infiltration from the surface (R) and pumping rates of irrigation wells (Q), on the RMSE of the transient simulation with respect to the calibration (Fig. 6) and the validation period (Fig. 7) of the flow model, as well as the calibration period (Fig. 8) of the transport model. The whole procedure includes the estimation of the RMSE variation due to the modification of the aforementioned parameters for ±5, ±10 and ±20 %.

Concerning the flow model, Figs. 6 and 7 show that the parameter of major influence on the RMSE is the pumping rate of irrigation wells, which appears to be more intense during the validation period of the model. Suffice to note that during this period a 20 % increase of pumping rates results to a RMSE variation equal to 130 %, while during the calibration period, the RMSE variation due to the same increase is 15 %. With regard to the other parameters, hydraulic conductivity and aquifer recharge have remarkable influence on the RMSE estimates, while specific storage is the parameter which is characterized by the smallest effect.
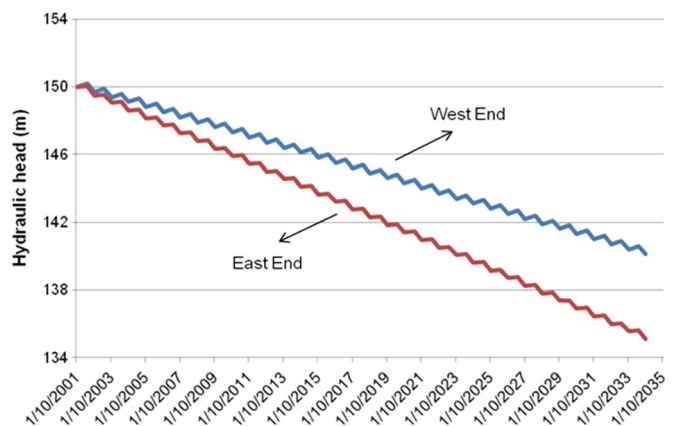

Fig. 5 Hydraulic head values at both ends of the aquifer's north boundary during the simulation period
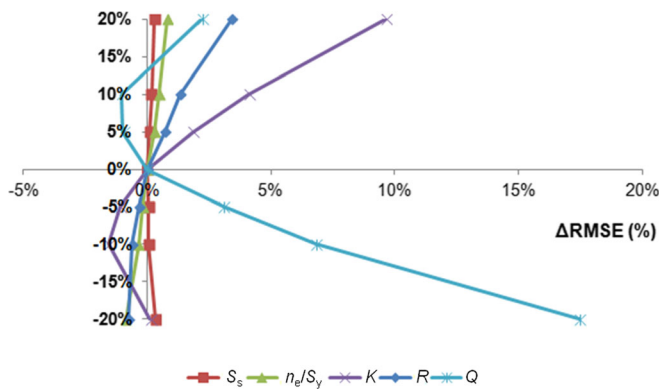
Fig. 6 Influence of the various model parameters (S_s specific storage, n_e/S_y effective porosity/specific yield, K hydraulic conductivity, R recharge, Q pumping rate) on the RMSE estimates during the calibration period of the flow model

Fig. 8 Influence of the various model parameters (S_s specific storage, n_e/S_y effective porosity/specific yield, K hydraulic conductivity, α_L longitudinal dispersivity, R recharge, Q pumping rate) on the RMSE estimates during the calibration period of the transport model

Figure 8 shows that, in the transport model, the parameter of major influence on the RMSE is the effective porosity. It should be noticed that a 20 % decrease of effective porosity results in a RMSE variation equal to 1,750 %, while a similar increase leads to a RMSE variation equal to 650 %. Concerning the other parameters, hydraulic conductivity and pumping rates have remarkable influence on the RMSE estimates, while specific storage is, as before, the parameter which is characterized by the smallest effect.

The main conclusion arising from the sensitivity analysis is related to the validity testing of the calibration procedure of the transient model, which is carried out by observing the RMSE variation due to the modification of the various model parameters. This is because if RMSE is decreased by modifying one parameter, the model calibration/validation is considered better that the initial one, whereas if RMSE is increased, the model calibration/validation is considered worse. Based on the aforementioned analysis and since, as shown in Fig. 6, the modification of specific parameters (e.g. hydraulic conductivity, pumping rates) during the calibration period of the flow model leads to the decrease of RMSE, these
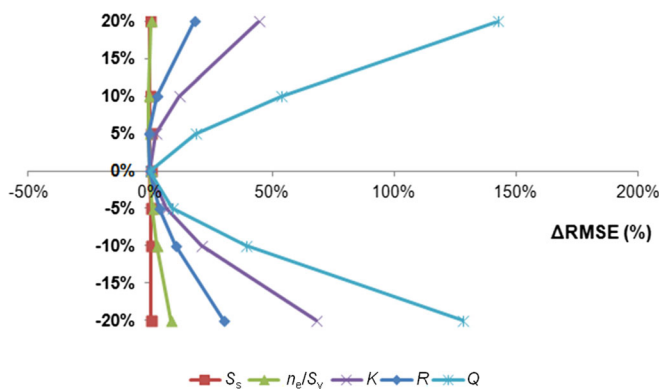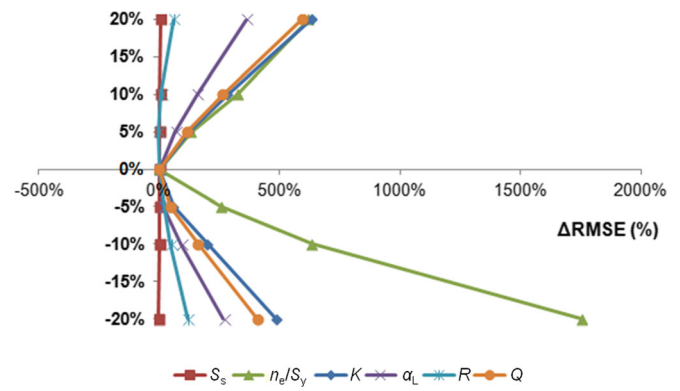
parameters should be readjusted and the model should be calibrated again following the procedure described in section 'Numerical modeling'. However, this was not done, because the aforementioned modifications in the case of both the flow model validation (Fig. 7) and transport model calibration (Fig. 8) result in an increase of RMSE to a greater extent than the flow model calibration. Therefore, the values assigned to the various model parameters lead to minor RMSE estimates rendering the calibration procedure quite satisfactory.

Study area

Groundwater flow

In Fig. 9, the spatial distributions of the hydraulic head (water level) at the beginning (Fig. 9a) and at the end (Fig. 9b) of the simulation period are displayed. It is obvious that high negative values of hydraulic head (in relation to the sea level) are observed in the southern and central part of the study area due to the intensive exploitation of groundwater resources in this region: approximately 75 % of the total abstractions is located in the aforementioned region that comprises most of the irrigated land and therefore most of the productive wells.

As a result, a large cone of depression is developed in the central part of the study area (centered at approximately 5 km from the coast) where the bulk of pumpage is accumulated. Moreover, the study area is characterized by a continuous decline of the water level over time, which on average amounts to 11.0 m for the whole simulation period (i.e. about 0.33 m/year).

In Fig. 10, the time variation of hydraulic head in two typical aquifer cells located in the most affected area is illustrated. In this figure, graph 1 corresponds to a cell without a well, while graph 2 refers to a cell with an irrigation well. The main conclusions arising from this figure center on (1) the continuous decline of the hydraulic head during the simulation period, (2) the seasonal fluctuation of the water levels due to the seasonal
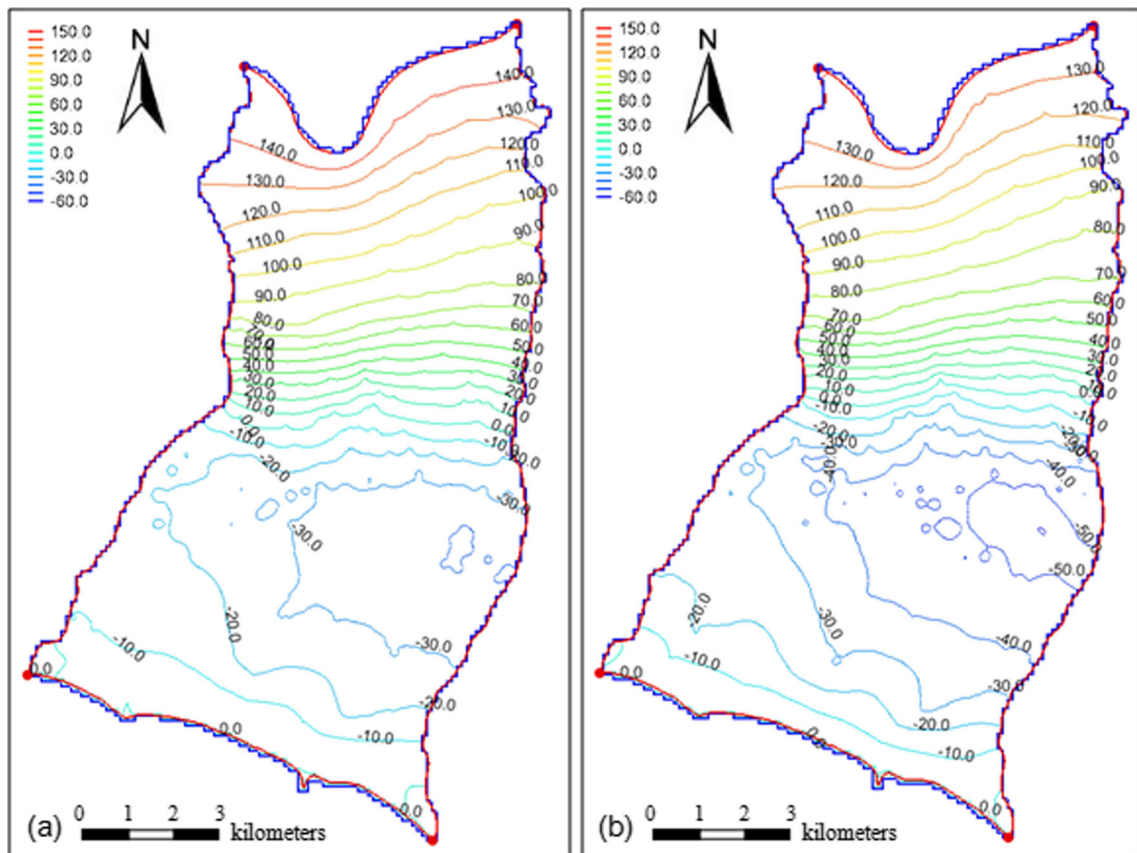

Fig. 7 Influence of the various model parameters (S_s specific storage, n_e/S_y effective porosity/specific yield, K hydraulic conductivity, R recharge, Q pumping rate) on the RMSE estimates during the validation period of the flow model

Fig. 9 Hydraulic head distribution **a** at the beginning (2001) and **b** at the end (2034) of the simulation period

operation of the irrigation wells (during the pumping season, 1st May–30th September) and (3) the different magnitude of fluctuations between the two cells, indicating the high influence of the well operation on the hydraulic head.

Regarding the flow budget, Fig. 11 shows the monthly values of the water balance components during the hydrological year "2002–2003", which coincides with the calibration period of the transient model. In this figure, the bars refer to the water inflow into the aquifer system, while the line corresponds to the outflow from the system. Moreover, the term "infiltration" is related to the total amount of water deriving from rainfall, irrigation return

Fig. 10 Hydraulic head variation in two typical aquifer cells during the simulation period

flows and leakages from water supply/wastewater networks. The main conclusion stemming from this figure is that the water outflow is higher than the water inflow during the pumping season (1st May–30th September), whereas the reverse happens during the non-pumping season (1st October–30th April). However, taking the whole year into consideration, the water outflow and thus the water demand is much more than the water supply.

Furthermore, Fig. 11 shows that in most months the aquifer recharge occurs mainly either from rainfall (November–March) or from irrigation return flows (May–September). However, the contribution of the water quantities entering the system through its north and south boundaries is equally important to the previous condition. In this case, what is worth noting is the amount of water entering the system from the south boundary, which is actually seawater.

In Fig. 12, the results obtained by the calculation of the annual water balance at different time periods of the simulation are presented (bars). It is far from clear that in all periods a net deficit in water balance is observed. Nevertheless, this deficit decreases over time, which is attributed to the ever-increasing amount of water entering the aquifer system from the south boundary, as well as to the fact that the other components of the water balance are either constant (i.e. infiltration water, pumping) or they
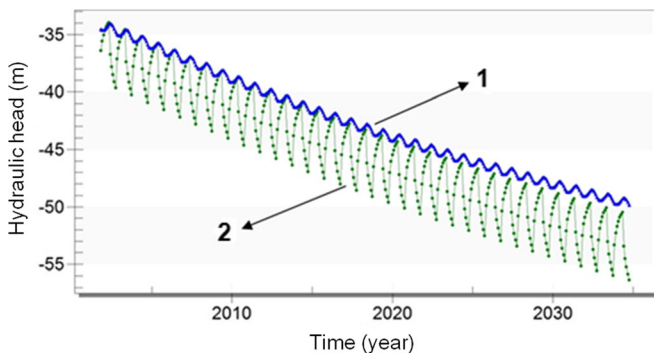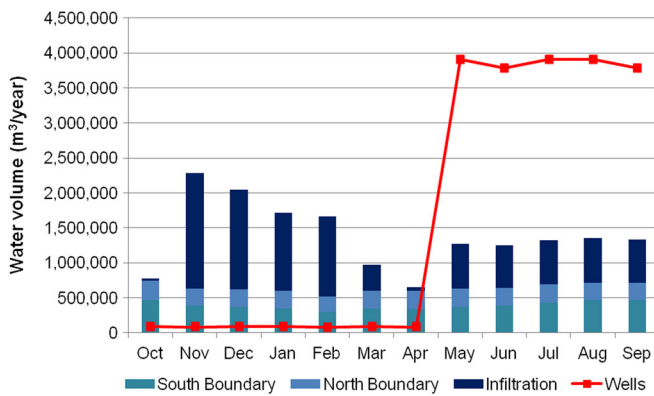
Fig. 11 Monthly distribution of the water balance components during the hydrological year of the transient model calibration

are reduced (i.e. north boundary) over time. In the same figure, in conjunction with the annual deficit of the aquifer at various time periods, the average water level decline (line) at the same time periods due to the aforementioned deficit is illustrated. As expected, the water level decline follows a downward trend over time similar to the water deficit.

Finally, in Fig. 13 the semi-confined portion of the aquifer at the beginning (Fig. 13a) and at the end (Fig. 13b) of the simulation period is illustrated (blue cells). As is obvious, semi-confined conditions occur mainly in areas located in the north and the south part of the aquifer, where water level is higher than the aquifer upper boundary; however, these areas shrink over time due to the continuous water level decline.

Seawater intrusion

In Fig. 14, the spatial distribution of chloride concentrations at the beginning (Fig. 14a) and at the end (Fig. 14b) of the simulation period is shown. As indicated, chloride concentrations increase and the seawater front expands over time, thereby causing the qualitative degradation of the aquifer to an increasing extent and rendering the local water reserves more and more unsuitable for use. In addition, according to these
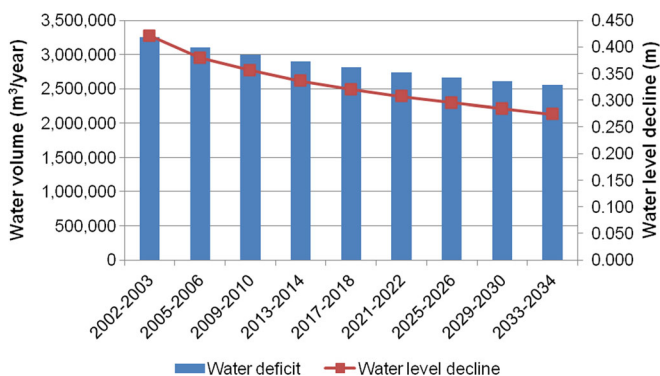

Fig. 12 Time variation of water deficit and water-level decline during the simulation period

figures, while the seawater front is at a maximum distance of 1.2 km from the coast at the beginning of the simulation (2001), it is projected that it will have penetrated about 2.2 km towards the interior of the aquifer by the end of the simulation (2034), thus covering a distance of 1 km during a period of 33 years.

In Fig. 15 the movement rate of the seawater front is presented by means of the movement rate of the 250 mg/L chloride concentration contour (1.3 % isochlor) at three different sections of the aquifer (sections A–A′, B–B′ and C–C′) based on its zonation (Fig. 4). It is clear that the distance of the seawater front from the coastline increases over time in all three sections, while it covers the largest distance in the case of section A–A′, mainly due to the high hydraulic conductivity value in zone 1 (Table 1). The same conclusions arise by estimating the average velocity of the seawater front movement during the simulation period for all three sections, which leads to the following results: A–A′ = 31.3 m/year, B–B′ = 22.0 m/year and C–C′ = 15.8 m/year.

Discussion

Methodology

The proposed methodology investigates the behavior of coastal aquifer systems under the absence of sufficient data and is based on the development of a series of numerical models: steady-state, false transient and transient models. Moreover, it attempts to determine the initial conditions for the transient model through a technique which is a simplification of the one described by Sanford and Pope (2010). The main advantage of the proposed technique is that, although available data are limited, it attempts to create a realistic salinity field for the transient simulation based on the available concentration measurements. Even though the models developed are quite simple and their calibration is not as rigorous as it should be, due to the lack of data, the proposed methodology creates a salinity profile which reflects the actual conditions quite satisfactorily. This is crucial for the model results and therefore for its use as a proper management tool. Furthermore, one more advantage of the proposed methodology is that it can be generally implemented in areas where basic information about the local geological, hydrogeological and hydrological conditions is available, since it was developed in order to compensate for the lack of sufficient data turning this lack into a prerequisite for the proper application of the methodology.

Provided that the proposed methodology includes the development of various numerical models, its evaluation and validity testing lies in evaluating the reliability and
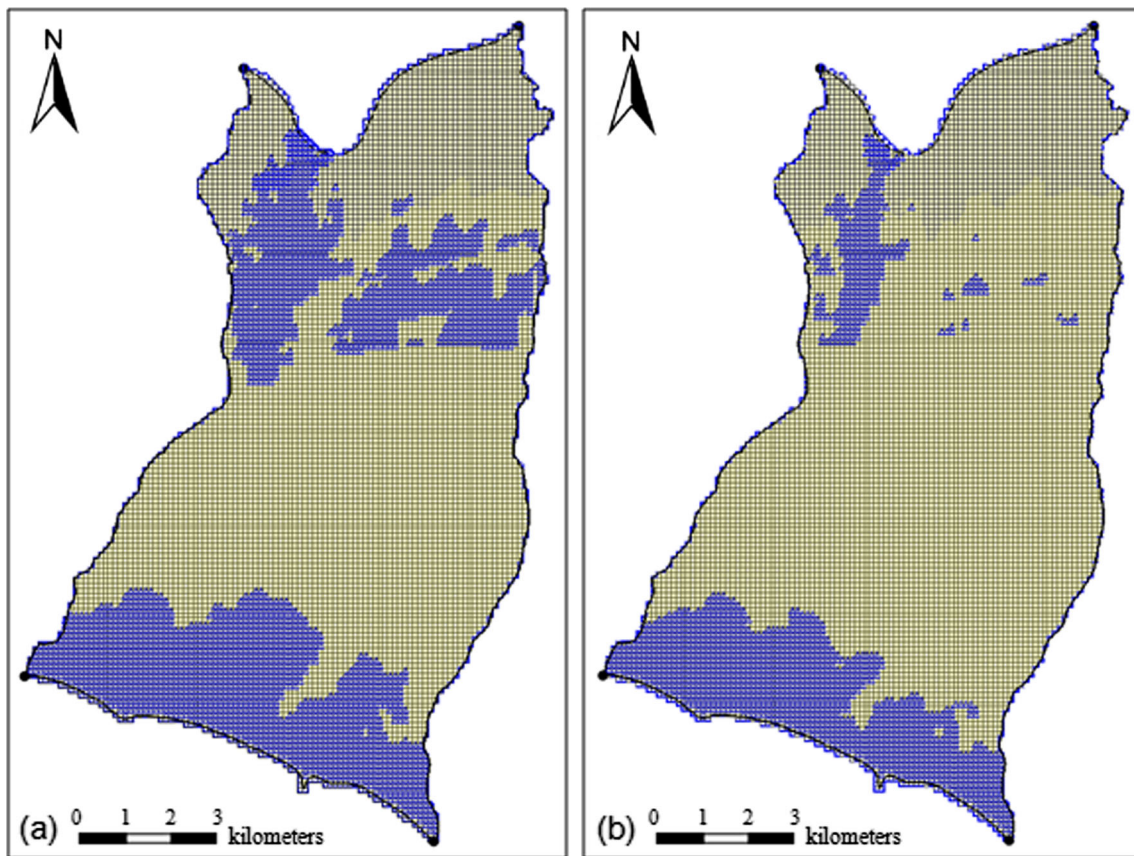
Fig. 13 Depiction of the semi-confined portion of the aquifer (*blue cells*) **a** at the beginning (2001) and **b** at the end (2034) of the simulation period

predictive capability of these models, which in turn are strictly linked to the evaluation of their calibration procedure. Therefore, as long as the calibration of the aforementioned models provides satisfactory results and minor error estimates, their reliability is increased and thus the robustness of the proposed methodology is enhanced. Therefore, according to the error estimates, the calibration procedure of all the models developed in this study is deemed completely satisfactory (MRE < 10 % in all cases). As a consequence, the simulations of both groundwater flow and seawater intrusion are considered quite valid, a fact that confirms the successful implementation of the proposed methodology.

In addition, the fact that modification and adjustment of the model parameters resulted in values contained within specific limits (determined either by previous research conducted in the study area or by the literature for similar types of sediments) magnifies the effectiveness of the calibration procedure and thus of the proposed methodology. Moreover, the robustness of the proposed methodology is further strengthened through the sensitivity analysis which was carried out in order to investigate the influence of the various parameters on the RMSE estimates. Through this procedure, optimal adjustment

of the various parameters was accomplished and simulation errors were minimized as much as possible, leading to the improvement of model reliability and predictive capability.

However, what is worth mentioning is that the proposed methodology is better adopted when there are no sufficient data, especially as far as the chloride concentrations are concerned. Selecting a specific time step of the false transient simulation, the results of which coincide with the observed values in the monitoring wells and are introduced as initial conditions to the transient simulation, is considered not feasible at all if more data exist. In that case, other techniques such as those described at the beginning (see section 'Introduction') , which are based on more available data, should be applied.

Study area

Groundwater flow: hydraulic heads

Simulating groundwater flow in the Nea Moudania aquifer affirmed the existence of high negative values (i.e. below sea level) of hydraulic head especially in the central part of
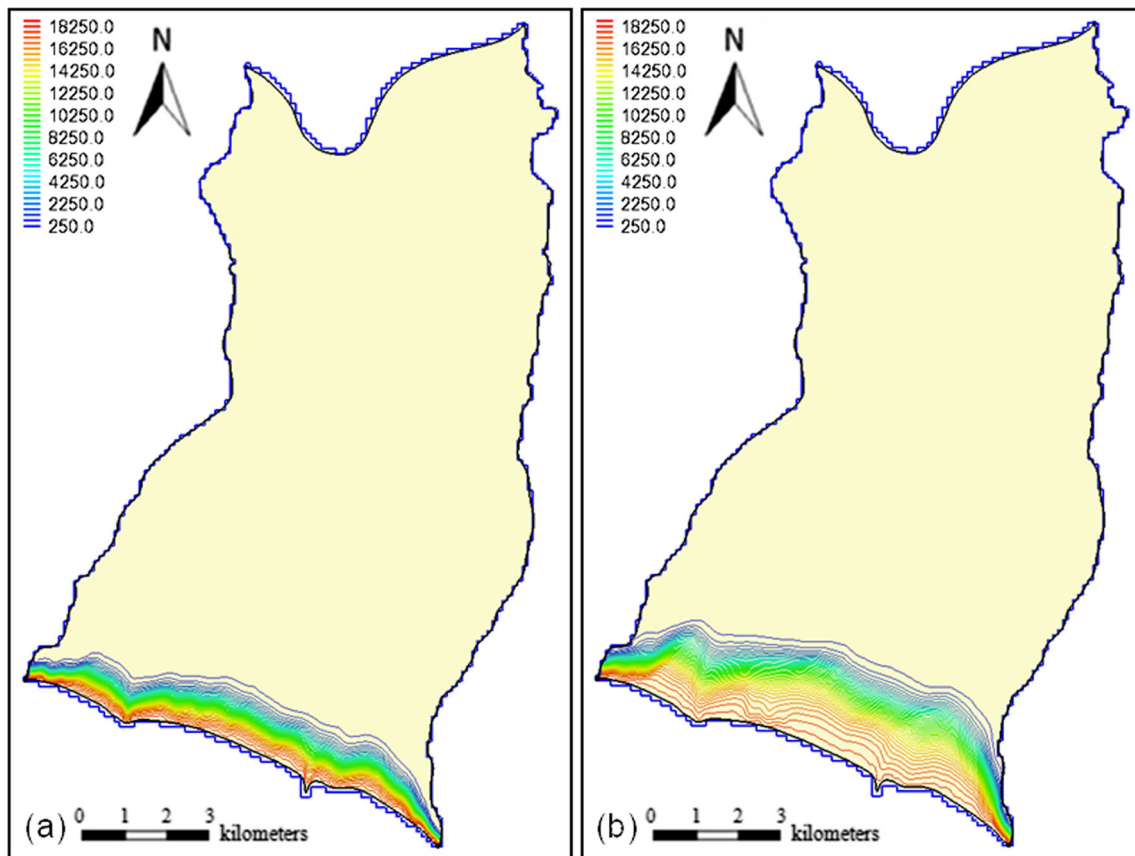
Fig. 14 Chloride concentration (in mg/L) distribution **a** at the beginning (2001) and **b** at the end (2034) of the simulation period

the region. These conditions have been observed since the beginning of the simulation period and they worsened over time, provided that all aquifer stresses (i.e. recharge and discharge components) were kept unmodified. Therefore, during the simulation period, a continuous water level decline takes place, which is directly linked to the negative water balance of the aquifer under study (i.e. a net water deficit). This ever-increasing water level decline has led to the reversal of the natural water flow and the intrusion of seawater, the volume of which increases progressively with time.

However, it should be mentioned that both the water deficit and the water level decline follow a downward trend over time, which is attributed to the fact that the amount of water entering from the sea increases, while the other components of the water balance are either constant (i.e. infiltration water, pumping) or they are reduced (i.e. north boundary) over time. In other words, the hydraulic head decreases over time but more slowly as the simulation progresses, since the water deficit is constantly decreasing. Nevertheless, when taking into consideration only the amount of freshwater (without taking into account the amount of water entering from the sea, since it is actually characterized as non exploitable), the water deficit appears

to be higher. Moreover, it shows a remarkable increase over time, thus leading to a significant reduction of freshwater reserves in the study area.

Seawater intrusion: chloride concentrations

Simulating seawater intrusion in the Nea Moudania aquifer resulted in the increase of chloride concentrations over time in the southern part of the aquifer, a fact that indicates movement of the transition zone towards the interior of the study area. However, this movement takes place at a different rate along the coastline, due to the zonation of the study area and the different values of the aquifer hydraulic parameters (e.g. hydraulic conductivity, effective porosity) assigned to these zones.

Conclusions

In the present study, in the case of a lack of adequate data, the numerical simulation of groundwater flow and seawater intrusion in overexploited coastal aquifer systems is attempted. To this task, a methodology applying the widely known codes MODFLOW, MT3DMS and SEAWAT is developed. The

Fig. 15 Movement rate of the 250-mg/L chloride-concentration contour (1.3 % isochlor) in three sections of the aquifer
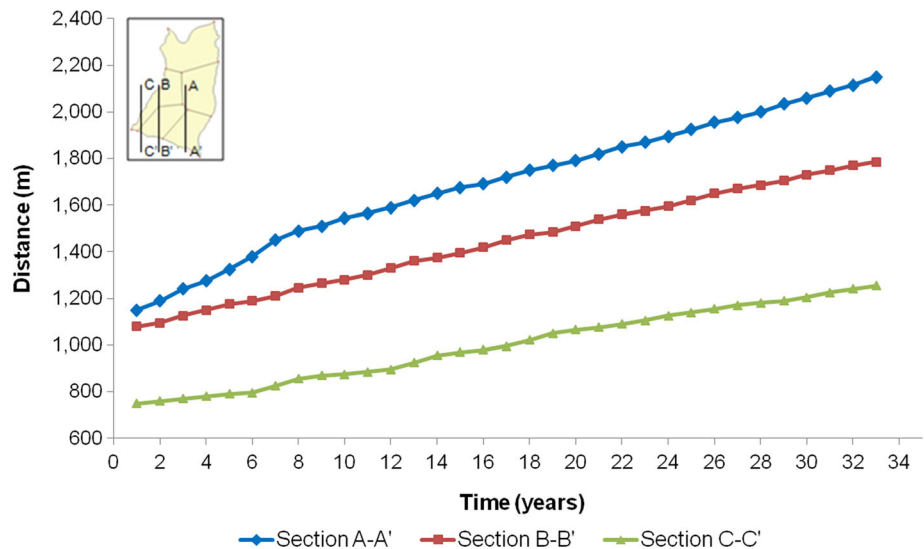
proposed methodology involves a series of numerical simulations, i.e. a steady-state, a false transient and a transient simulation, which are linked together in order to achieve the best possible adjustment of the various model parameters, i.e. hydraulic conductivity, storativity, effective porosity, longitudinal dispersivity, aquifer recharge, pumping rates, as well as to get the initial conditions for the transient simulation.

The validity of the proposed methodology is based on the reliability of the aforementioned models, which is assessed through their calibration procedures and the estimation of the simulation errors (ME, MEA, RMSE, MRE). According to this statement and since the error estimates are low, indicating a quite satisfactory calibration procedure, the proposed methodology is considered quite valid. Moreover, the robustness of the proposed methodology is enhanced through a sensitivity analysis which refers to the transient model and is conducted in order to minimize the simulation errors and thus maximize the reliability and predictive capability of the model, by modifying and properly adjusting the various model parameters. Through this procedure, the parameter values leading to the best possible calibration with the conceptual model used and the hypotheses made are obtained. Furthermore, it is concluded through the sensitivity analysis that the flow model is very sensitive with respect to changes in pumping rates, while the transport model is very sensitive with regard to changes in effective porosity. In both cases, specific storage is the aquifer parameter with the smallest effect on the model results.

With regard to the study area, the simulation of groundwater flow indicates a significant quantitative degradation of the aquifer system, since an annual water deficit is observed due to the overexploitation conditions prevailing in the whole region in order to meet the increased water needs. The hydraulic head decreases over time, while it lies well under the mean sea level in the central and southern part of the region, a fact that

leads to the development of a large cone of depression centered at approximately 5 km from the coast. These conditions cause the intrusion of seawater through the southern boundary, which is in hydraulic connection with the sea. More specifically, as is concluded from the simulation of seawater intrusion, the transition zone constantly advances towards the interior of the aquifer system, causing its progressively increasing qualitative deterioration.

In order to alleviate the aquifer quantitative and qualitative degradation immediate measures should be taken. The following are among the most critical measures that could be suggested: (1) reducing the pumping rates of the irrigation wells, (2) modifying the general pumping scheme of the study area by redistributing the location of the irrigation wells and especially those sited in the most affected area, (3) using surface runoff as an alternative source of irrigation water, (4) constructing artificial recharge wells in close proximity to the coast and (5) adopting specific practices aiming to preserve water (e.g. implementing irrigation practices with low water usage, improving the irrigation program by taking into account parameters such as effective rainfall and crop water needs, charging the use of water or buying groundwater rights from the farmers). The implementation and evaluation of these measures could be based on the transient model developed in this study in conjunction with a well-organized monitoring system. If no management measures are taken and the current trend of groundwater abstraction continues, the quality of fresh groundwater will be even more degraded, while the agricultural land will shrink and the crop yield will be reduced significantly. In other words, the apparent results of a likely inaction would cause negative environmental and socio-economical effects to the study area.

which significantly improved the manuscript. This work was carried out within the framework of a PhD thesis conducted by Ilias Siarkos. Assistance for field data collection was provided by the Municipality of Nea Propontida, Halkidiki.

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