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# Screening of sustainable groundwater sources for integration into a regional drought-prone water supply system

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## Abstract

This paper reports on the qualitative and quantitative screening of groundwater sources for integration into the public water supply system of the Algarve, Portugal. The results are employed in a decision support system currently under development for an integrated water resources management scheme in the region. Such a scheme is crucial for several reasons, including the extreme seasonal and annual variations in rainfall, the effect of climate change on more frequent and long-lasting droughts, the continuously increasing water demand and the high risk of a single-source water supply policy. The latter was revealed during the severe drought of 2004 and 2005, when surface reservoirs were depleted and the regional water demand could not be met, despite the drilling of emergency wells.

For screening and selection, quantitative criteria are based on aquifer properties and well yields, whereas qualitative criteria are defined by water quality indices. These reflect the well's degree of violation of drinking water standards for different sets of variables, including toxicity parameters, nitrate and chloride, iron and manganese and microbiological parameters. Results indicate the current availability of at least 1100 $\mathrm{l\,s^{-1}}$ of high quality groundwater (55% of the regional demand), requiring only disinfection (900 $\mathrm{l\,s^{-1}}$) or basic treatment, prior to human consumption. These groundwater withdrawals are sustainable when compared to mean annual recharge, considering that at least 40% is preserved for ecological demands. A more accurate and comprehensive analysis of sustainability is performed with the help of steady-state and transient groundwater flow simulations, which account for aquifer geometry, boundary conditions, recharge and discharge rates, pumping activity and seasonality. They permit an advanced analysis of present and future scenarios and show that increasing water demands and decreasing rainfall will make the water supply system extremely vulnerable, with a high risk of groundwater salinization and ecosystem degradation.

# 1 Introduction

## 1.1 The need of integrated water resources management

The concept of integrated water resources management (IWRM) has gained importance worldwide, and the need to deal with its complexity has led to the increased use of decision support tools. Several articles dealing with such tools have been published, e.g. for integrated river basin management to simulate the economic impacts of policies in drought periods (Characklis et al., 1999; Booker et al., 2005), to quantify the economic value of stream flow (Pulido-Velazquez et al., 2006, 2008), for policy options and water allocation (Letcher et al., 2004), for trade-offs between competing uses (Burke et al., 2004), for analysis of the impact of climatic changes (Tanaka et al., 2006), and for trade-offs between efficiency, equity and sustainability in the design of water programs (Ward and Pulido-Velazquez, 2008).

IWRM drives individual sectors to coordinate actions and collaborate with each other and enhances stakeholder participation, transparency and cost effective local management (Mylopoulos and Kolokytha, 2008). The objectives of IWRM for socio-economic development and sustainable development require the adoption of three key policy principles (Postel, 1992): i) equity, i.e., water is a basic need and no human can live without a minimum amount and quality; ii) ecological integrity, i.e., water resources are only sustainable if the environment is capable of regenerating water of sufficient quality and quantity; iii) economic efficiency, i.e., water is a scarce resource, though not necessarily an ordinary economic good (Savenije, 2002), therefore must be used efficiently.

In the Algarve region, south Portugal, there are a number of factors that impose the need for an IWRM scheme. First of all, in semi-arid regions such as the Algarve, the seasonal and annual variations in rainfall are extreme, posing serious challenges for water supply planning and management. Second, the intensity and frequency of occurrence of extreme droughts will most likely increase significantly in the future, according to the climate change study of Portugal (Santos and Miranda, 2006). Climate models predict a 10 to 40% lowering of rainfall in the south of Portugal by the end of this century. In general, climate change will be pronounced in the Mediterranean region, due to the magnitude of expected changes in temperature and rainfall patterns (Giorgi, 2006). Moreover, water resources are arguably the most important domain to be considered in studies that assess the impact of climate change (Cunha et al., 2006). This importance stems from the fact that climate change has direct impacts on the availability, timing and variability of water supply and demand, and is also related to the significant consequences of these impacts on many sectors of our society.

Third, the water demand in the region is continuously increasing, owing to the growth of population and tourism. Agriculture is still by far the major water-consuming activity. Annual water consumption in the Algarve is currently estimated to be around 280 $hm^3$ ($\times 10^6$ $m^3$), of which 69% is for agriculture, 23% for public supply and 8% for other activities (golf courses, industry, private wells). Though groundwater is the main source for irrigation (165 $hm^3$), irrigation with surface water is gaining importance.

Fourth, a water supply policy based on a single source significantly increases the risk of failing to meet water demand, either during long drought spells or due to accidental events. Such failure occurred in the Algarve during the severe drought of 2004 and 2005, when stored water volumes in surface reservoirs dropped dramatically, reaching the exploitation limit in the west. Currently, an additional dam (Odelouca) is being built, which will be completed in 2010, but studies have shown that even then water demand will be hard to meet in the long term (Hidroprojecto and Ambio, 2005).

### 1.2 Developing an integrated water resources management scheme

For all the above-stated reasons an IWRM scheme for public water supply is essential to guarantee a stable and sustainable public water supply, obtaining a correct balance between the various water resources. To accomplish those tasks, a decision support system based on an optimization model is currently under development, in the scope of the R&D project ("OPTEXPLOR"), under contract between the team and the Water Utility Águas do Algarve, S.A. to support decisions about: i) different sources to be mobilized, namely surface water, groundwater, desalinated seawater, and treated wastewater – the latter only for irrigation; ii) the operating policy of such infrastructures for different scenarios (climatic and operational) and iii) supplying water of very good chemical and organoleptic qualities.

Future developments may come to include: i) the location and the size of water intake and water distribution infrastructures; ii) the location, size and technologies for water treatment, wastewater treatment and water desalination infrastructures.

The optimisation decision model being developed aims to promote an efficient and sustainable water use at the basin scale and assumes the form of an integrated framework to assess water management policies. The feasibility of the solutions and the sustainability of the system being modelled are guaranteed with the formulation of an adequate objective function and the introduction of the necessary constraints. In its objectives the Water Utility looks to minimize intervention costs, while satisfying the system demand and providing water with the appropriate quality for human consumption. Surface water storage in dams is modelled with mass balance equations. The groundwater flow in the aquifers is represented by means of distributed parameter models and the use of the matrix response approach (e.g. Maddock, 1972). Water quality is specified in terms of the volumetric ratio of water from different sources (Yang et al., 2000). The model also takes into account prior decisions taken by the high-level Administration about the distribution of water volumes stored in surface reservoirs for both public water supply and irrigation. Minimum piezometric heads, maximum abstraction from wells and minimum discharges from dams for ecosystem preservation are considered sustainability criteria, and are included in the optimisation decision model.

### 1.3 Objectives of the article

Under an IWRM scheme, naturally, groundwater will be a valuable resource for public supply. Before the existence of the MPWSS, public water supply was mainly supported by groundwater. Moreover, several studies have pointed out the advantage of the joint use of surface and groundwater for treatment requirements (e.g. Campinas et al., 2001). Since well yield and water quality vary significantly over space and time, one of the necessary tasks of the OPTEXPLOR project was the quantitative and qualitative screening and selection of public groundwater wells for integration into the MPWSS. Only the best wells should be used for public supply and included in an integrated management scheme of the water resources. This paper aims to present the methodology and the results of the selection made, as well to analyze the sustainability of groundwater withdrawals from the screened wells. This latter analysis is essential as overexploitation can lead to degradation of the aquifer (e.g. due to seawater intrusion) and desiccation of associated wetlands. There are no general rules to define sustainable yields, because they depend on regional factors such as climate (and climate change), the hydrogeological setting, the particular location of the wells, the presence of groundwater dependent ecosystems, as well as social, economic and legal aspects (Sophocleous, 2000; Custodio, 2002; Alley and Leake, 2004; Maimone, 2004; Seward et al., 2006; Ponce, 2007). Therefore, regional groundwater flow models are used to simulate the response of the aquifers to different rates of groundwater pumping. In this paper, flow simulations are performed in the Querença-Silves aquifer system, the largest and most productive system of the Algarve.

## 2 Study area

### 2.1 Climate

The Algarve region (5400 km$^2$) is the southernmost province of Portugal, as indicated in Fig. 1. The region is characterized by a warm Mediterranean climate. A mean annual precipitation of 653 mm was calculated for the period 1941/42–1973/74 (Loureiro and Nunes, 1980). The precipitation regime is irregular, having intermittent periods with short and sharp floods in the winter and a long dry period in the summer. In addition, there may be extreme events such as inter-annual periods of drought.

## 2.2 Hydrogeology

The first hydrogeological characterization of the Algarve was presented in the work of Trac (1981). The work describes the Palaeozoic unit consisting of metamorphic rocks covering an area of about 3700 $km^2$ in the north, and nine aquifer systems mainly supported by Jurassic, Miocene and Quaternary formations, covering an area of 1700 $km^2$ in the coastal Mesocenozoic strip. A detailed water balance is presented, considering the separate estimation of the surface and groundwater components. Other detailed studies followed (e.g. Silva, 1984, 1988; Almeida, 1985), dealing with main aquifer dynamics and geochemistry.

The actual state of development of the Algarve hydrogeology allows the definition of 17 aquifer systems with regional importance, occupying 1074 $km^2$ of the referred Mesocenozoic strip. The definition of these 17 aquifer systems was proposed by Almeida et al. (2000) and their location and geometry are shown in Fig. 1. The most productive aquifers are built up of karstified limestones and dolomites. Table 1 presents the main characteristics of each system, including dominant aquifer lithology, average yield and annual recharge. The six most important aquifers for public water supply are M2, M3 (northern sector), M5 (also significantly exploited for irrigation), M8, M9 and M14. Of the other aquifer systems, M7, M10 and M12 are particularly important groundwater suppliers for irrigation. Due to its large area and significant recharge, as well as the high degree of karstification, aquifer system M5, known as Querença-Silves, constitutes the most important groundwater reservoir in the Algarve Region (see Fig. 1). Its geometry is pictured in Fig. 2. Built up of massive karstified limestones and dolomites, the system is bounded to the north by clays, mudstone and evaporates and to the south by a large thrust-fault zone, forming groundwater flow barriers. Groundwater discharge occurs at the springs located along the aquifer boundary, but the main discharge area is in the west, near the Arade estuary (Fig. 2). Important and sensitive surface/groundwater ecotones and associated groundwater dependent ecosystems exist at the location of these springs, many of them classified as protected areas.

### 2.3 Evolution of the use of groundwater through time

Previous publications (Monteiro and Costa, 2004; Nunes et al., 2006a; Stigter et al., 2007) provide an overview of the evolution of surface and groundwater use in the Algarve. Until halfway through the 20th century groundwater was exploited for domestic use and agriculture through hand-dug wells with large diameter and shallow depth. An exponential rise in water demand occurred in the 1960s, mainly associated to the expansion of irrigated agriculture and the growth of tourism. This was possible thanks to the introduction of drilling technologies that allowed the construction of thousands of boreholes in the region, without adequate regional planning nor supported by existing knowledge of the region's hydrogeology.

Despite the construction of the Arade and Bravura dams, groundwater remained the dominant source for irrigation and public supply until the end of the 20th century. Public supply was locally and independently managed by each of the 16 municipalities (whose wells are located in Fig. 1), so that regional water supply planning, management and quality control was impossible. At the same time, a few aquifers were suffering increasing pressures from human activities, such as nitrate contamination in irrigated areas (Stigter et al., 1998, 2006a, b) and saltwater intrusion in coastal sectors (Carreira, 1991, Salgueiro and Ribeiro, 2001).

The efforts to abandon groundwater as a source for public supply started in 1998, after a large investment in new infrastructures (Funcho and Odeleite dams, water treatment plants, regional distribution system) and in the rehabilitation of others (Beliche and Bravura dams). The abrupt policy change led to the implementation of a multi-municipal public water supply system (MPWSS) entirely based on surface water supplied by the reservoirs, replacing the individual municipal systems. In 2002 more than 80% of the total public supply was sustained by surface water, as illustrated in Fig. 1. The major benefits of the use of surface water were felt in water quality control, which proved to be much easier in a MPWSS.



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However, the negative consequences of this single-source water supply policy were felt during the severe drought that occurred in 2004 and 2005, causing an overall 87% depletion of the surface water storage and the complete depletion of the Funcho and Arade reservoirs. Restrictions on water use were imposed for all economic sectors, but particularly agriculture, with exceptional measures being taken to avoid the total disruption of the public water system. The Regional Water Utility (AdA) decided to drill emergency wells in the Querença-Silves aquifer system (M5). Despite the additional 11 $hm^3$ supplied by this aquifer, the water demand could not be met by the MPWSS and formerly abandoned municipal wells had to be reactivated (for location see Fig. 1). In total 42% of the public water supply was supported by groundwater in 2005. An overview of the distribution of groundwater supply among the main aquifer systems is shown in Fig. 1, revealing the dominant role of M5.

## 3 Well screening and selection

### 3.1 Methods

The screening and selection of groundwater wells for integration into the MPWSS was based on quantitative and qualitative criteria. Regarding the former, the overall characteristics of the regional aquifer systems in the Algarve were studied based on published reports (e.g. Silva, 1984; Silva, 1988; Almeida et al., 2000) and well yields were analyzed individually when available. A well yield criterion of 15 $l\,s^{-1}$ was defined, based on field experience and expertise and met by three-quarters of the wells with known yields. Some wells with lower yield were included, depending on their location and on the aquifer properties.

The qualitative screening procedure was based on chemical analyses performed in the period 1995–2005 for the city councils and by the Regional Environmental Agency. In total 279 wells were included in the present study, of which 110 are municipal wells for public supply and the remaining are private wells, drilled mostly for irrigation and domestic use. The latter cannot be selected for public supply, but were included to provide a regionally more comprehensive overview of groundwater quality in the Algarve.

Based on all the available analyses, a standard violation index (SVI) was calculated for each well, for different sets of variables, based on the associated drinking water standards, represented by the parametric values (PVs) of the EU Drinking Water Directive (98/83/CE). Such water quality indices (WQIs) have been developed with the aim of rapidly combining a large quantity of chemical information of a water sample into a single value and thereby easily monitor spatial and temporal fluctuations of water quality (e.g. Harkins, 1974; Backman et al., 1998; Stigter et al., 2006b). For a given period $t_0 - t_u$, where $t_0$ and $t_u$ are the initial and final dates of the considered analyses, the SVI is calculated in the following way:

$$\mathrm{SVI}_{\mathrm{vars},i} = \frac{NV_{\mathrm{vars},i}}{NA_{\mathrm{vars},i}} \quad (1)$$

where, $\mathrm{SVI}_{\mathrm{vars},i}$ is the standard violation index of the variable set "vars" in well $i$, $NA_{\mathrm{vars},i}$ the number of analyses and $NV_{\mathrm{vars},i}$ the number of drinking water standard violations. The created SVIs, shown in Table 2, vary between 0 and 1 (100% of violations).

$\mathrm{SVI}_{\mathrm{Tox}}$ is based on toxic substances. Any violation revealed by this index results in the exclusion of the corresponding well for drinking water purposes and requires further attention. $\mathrm{SVI}_{\mathrm{Microb}}$ is an indicator of microbiological contamination and the level of disinfection required during water treatment. As this type of groundwater contamination is frequent, it is useful to classify the water samples among the water treatment categories A1, A2, A3 defined by Council Directive 75/440/EEC (for surface water, but also applicable to groundwater). $\mathrm{SVI}_{\mathrm{NO_3,Cl}}$ combines two parameters that are typical indicators of a disturbance of the natural state of groundwater, often due to human activities. For instance, agricultural practices have a large impact on concentrations of both parameters in the region, due to excess fertilization and groundwater extraction for irrigation (e.g. Stigter et al., 2006a,b). Neither nitrate nor chloride are removed from water in conventional treatment plants, hence violations of drinking standards impair the use of the well for this purpose, though groundwater may be mixed with surface water to obtain acceptable levels. $SVI_{Fe,Mn}$ indicates the need of iron and manganese removal from groundwater for public supply, increasing the treatment costs. Both parameters are treated at water treatment plants, though they can form stable complexes with humic substances and become more resistant to oxidation.

For the screening and selection procedure, the wells were distributed among six quality classes. Class 1 represents the highest quality and includes wells extracting groundwater that after a basic disinfection are ready to be used for drinking water purposes. The selection criteria for the first class were defined as follows:

1. all samples of the well in class A1 of required microbiological treatment (75/440/EEC);
2. $SVI_{Fe,Mn} \leq 0.25$ (maximum one violation in four analyses);
3. $SVI_{Tox}=0$ and $SVI_{NO_3,Cl}=0$ (no violations permitted).

The reason of loosening the $SVI_{Fe,Mn}$ criterion is twofold. First, elevated concentrations in groundwater are caused by natural rather than human factors. Second, concentrations of iron and manganese are reduced during water treatment. Based on the same reasoning, the hypothesis of loosening the criterion for $SVI_{NO_3,Cl}$ was rejected: high concentrations of $NO_3$ and Cl are due to human impact in the majority of cases, whereas their removal during water treatment does not occur.

Wells of class 2 and 3 only require additional iron and manganese removal ($SVI_{Fe,Mn}>0.25$) and disinfection (class A2 and A3), respectively, whereas wells of class 4 require both. Hence, wells of classes 2, 3 and 4 can all be used in the MP-WSS as long as extracted groundwater is previously diverted to water treatment plants or receives the required treatment in situ. The last two classes include wells that are not considered for drinking water, either due to the presence of nitrate and chloride (class 5) or toxic substances (class 0).

### 3.2 Results and discussion

Of the referred parameters, it is widely recognized that microbiological contamination can have serious consequences for public health. Nitrate in drinking water is generally considered to constitute a health hazard for babies and young infants. Its toxicity is mainly attributable to the reduction to nitrite and associated to methaemoglobinemia, although several recent studies question this fact (e.g. L'hirondel and L'hirondel, 2002; Addiscott, 2006). As far as known, no health problems are related to iron and manganese for humans, but their presence in public water supplies can create serious problems. In distribution systems, they can cause difficulties by supporting the growth of iron bacteria and impart a taste to water (for iron) detectable at very low concentrations (Sawyer et al., 2003).

The spatial distribution of one of the standard violation indexes, $SVI_{NO_3,Cl}$, in the main aquifer systems of the Algarve is shown in Fig. 3a. Since only municipal wells with zero index values are considered for integration into the MPWSS, the map already provides a good overview of the aquifer systems that are most interesting from a qualitative point of view. The highest index scores (worst water quality) are observed in the coastal aquifer systems M6, M12 and M15. Data analysis reveals that in aquifer systems M6 chloride dominates the $SVI_{NO_3,Cl}$ and that mostly municipal wells are affected. This proves that groundwater extraction has led to saltwater intrusion, the origin of which cannot be specified without further study. Other affected areas are the eastern sector of M2 and the southern sector of M3. On the contrary, wells in aquifers M12 and M15 are mostly private and high scores are related to both nitrate and chloride. The principal diffuse source of nitrate is the use of mineral fertilizers in agriculture. In addition, irrigation with locally extracted groundwater induces a groundwater cycle that gradually increases the nitrate concentrations along with the overall groundwater salinity, especially in semi-arid regions such as the Algarve. The problem has been officially recognized for aquifer systems M12 and M15, as they include the only two nitrate vulnerable zones designated in the Algarve in compliance with the Nitrates Directive

91/676/EEC (Stigter et al., 2006b).

The lowest $SVI_{NO_3,Cl}$ scores (best water quality) are found in aquifer systems M2 (western sector), M3 (northern sector), M5 (except for its westernmost sector, near the Arade estuary), M8, M9 and M14. These are carbonate aquifers, with high recharge rates and groundwater velocities, and often located more inland, protected from seawater intrusion. Their dominant role for public water supply is manifested in the map of Fig. 3b, which locates the wells selected for integration into the MPWSS, as a result of the screening procedure.

Figure 4a shows the distribution of the selected wells distributed among the six treatment classes, according to the selection criteria. Over 60 municipal wells can be integrated into the MPWSS, of which 42 belong to class 1, i.e. only require disinfection prior to their use for drinking water. The private wells present an overall lower groundwater quality, mainly attributable to high $SVI_{NO_3,Cl}$ scores. The presence of toxic substances (i.e. lead, arsenic, nickel, mercury and pesticides) was only detected in 15 of the 5670 analyses. Lead was found twice in the same well in a short period of time. Nickel was found in six samples, but only on two different dates, whereas mercury was found in two samples on the same date. These cases seem to indicate the occurrence of local accidental spills, without any long-term consequences. In fact, three municipal wells with a $SVI_{Tox}>0$ and therefore not considered for public supply in this study, are currently still operational, located in the largest aquifer system (M5) and with a total well yield of $111\,l\,s^{-1}$.

Figure 4b presents the cumulative yield from municipal wells for the six main aquifer systems. Although several other aquifer systems contain wells that passed the screening process, it was decided to leave them out of the calculations, due to generally lower aquifer productivity (M4, M10) or problems with water quality in wells located nearby in the same aquifer (M6, M12). It can be observed that almost 900 l/s can be obtained from groundwater after disinfection, whereas an additional 400 l/s can be included after iron and manganese removal and additional disinfection. These numbers are based on maximum installed pumping capacity and continuous pumping and are of little use for water supply management, because they do not consider the limiting factors for groundwater availability, i.e. natural recharge, the presence of other water consumers (mainly agriculture) and ecological demands. The next section will deal with the sustainability of groundwater withdrawals from the selected wells.

## 4 Sustainability assessment of public water supply from selected wells

Safe yield is defined by Sophocleous (1997) as the attainment and maintenance of a long-term balance between the amount of groundwater withdrawn annually and the annual amount of recharge. Currently, the emphasis has shifted to sustainable yield (Sophocleous, 2000; Custodio, 2002; Alley and Leake, 2004; Maimone, 2004; Kalf and Woolley, 2005; Seward et al., 2006; Ponce, 2008), which reserves a fraction of safe yield for ecological demands. This need was already recognized a long time ago by Lee (1915) and Theis (1940), but no general rule exists as to what percentage of recharge can be considered sustainable. The latter depends on many regional factors, such as climate (and climate change), hydrogeological setting, the particular location of the wells and the presence of groundwater dependent ecosystems.

Table 3 and Fig. 4b present a first impression of the relation between the screened installed yield from municipal wells and the mean annual recharge ($R_n$). Two scenarios of maximum abstraction rates are considered here: 100% and 60% of $R_n$. Hence, only the latter scenario accounts for a long-term groundwater discharge component (ecological demand), but can also be seen to reflect reduced recharge caused by climate change. Climate models predict a maximum 40% lowering of precipitation by 2100 in the south of Portugal (Santos and Miranda, 2006). In order to assess available groundwater volumes for public supply, other water consumers need to be included, which is done for agriculture, by far the most significant consumer in the region (see Table 3).

The curves in Fig. 4b allow a rapid perception of the available well yields requiring disinfection (class 1) or further treatment (class 2–4) and those yields limited by the imposed criteria of sustainability. The four curves are only distinguishable for the



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smaller aquifer systems M8, M9 and M14, clearly indicating that abstraction rates are limited here by natural recharge. In the safest scenario, i.e. that of 40% groundwater discharge, maximum available well yields are between 30% and 50% of installed yield, and even in the "no discharge" scenario maximum yields are 75% and 82% of installed yield for M9 and M8, respectively.

Nevertheless, their contribution can be significant. For instance, groundwater of aquifer system M14 and the northern sector of aquifer system M3 can be diverted to the nearby located treatment plants, where they can be mixed with surface water prior to treatment. The typical natural characteristics of groundwater, such as high degree of hardness and low turbidity, make it extremely advantageous for mixing with surface water, with regard to treatment requirements (e.g. Campinas et al., 2001). In order to assess water quality changes driven by mixing hydrochemical models are being applied.

Even when considering that 40% of $R_n$ is unavailable (due to a combination of reduced recharge and ecological requirements), the total available yield from municipal wells is 35 $hm^3$ (Table 3), more than 1100 $l\,s^{-1}$ or 55% of the regional water demand in the Algarve (Fig. 4b). With an installed well yield of 21.5 $hm^3$ based on the screening procedure, aquifer system M5 is by far the largest public groundwater provider. Studying its response to such large withdrawal rates is therefore a critical task, which can be supported by a groundwater flow simulation model. Such a model can simulate groundwater flow under natural conditions and subsequently incorporate pumping activities from the screened wells, as well as from private irrigation wells, so that the risk of overexploitation can be comprehensively assessed for different scenarios. Overexploitation in aquifer system M5 could lead to the drying up of springs and associated wetlands, as well as aquifer degradation due to seawater intrusion.



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## 4.1 Groundwater flow model of aquifer system M5

### 4.1.1 Methods

The first step towards synthesizing the aquifer system at the regional scale using a numerical model involved the transformation of the available hydrogeological information into a general conceptual flow model, considering: (1) the geometry of the aquifer system, (2) the boundary conditions and location of the main discharge areas and (3) the recharge and discharge rates (Monteiro et al., 2006, 2007). The geometry of the aquifer system is pictured in Fig. 2 and discussed in Sect. 2.2. Boundary conditions were defined as constant head along the Arade estuary in the west and no-flow for the remaining part. Since there is strong tidal influence in the estuary, it would be more accurate to define a specified (non-constant) head boundary condition, which will be tested in future model runs. Boundary conditions could have been defined for several springs in the central and eastern sector, but model variants which included this step, revealed a minor impact on the regional flow pattern and water balance. Annual recharge rates, depicted in Fig. 2, were determined as a mean percentage of deep infiltration of precipitation, based on the method of Kessler (1965) and semi-empirical formulae of Thornthwaite (1948), Coutagne (1954) and Turc (1954). Accurately interpolated data of Nicolau (2002) were used to characterize the spatial distribution of rainfall.

The defined conceptual flow model was translated to a finite element mesh with 11 663 nodes and 22 409 triangular finite elements. For steady-state simulations, the direct solution was implemented using a standard finite-element model based on the Galerkin method of weighted residuals. Preliminary results showed that the hydraulic parameters obtained from pumping tests were not adequate to obtain realistic simulations of the spatial distribution of hydraulic head in several sectors. Therefore, transmissivity (T) was estimated by inverse modelling. Calibration was performed using the Gauss-Marquardt-Levenberg method, implemented in the nonlinear parameter estimation software PEST (Doherty, 2002). Optimization of the results proved a timeconsuming procedure, based on a thousand model runs, where variants were tested to search for the best reproduction of the equipotential surface, meeting the water balance criteria. In total 23 T zones were defined, where the behaviour of piezometers allowed a reasonable fitting of field data using a single value of T. Figure 5 presents their spatial configuration and the resulting plot of calculated versus observed heads, showing a satisfying fit with a calculated mean relative error of 16%.

In the current analysis, three scenarios were created:

SCENARIO 1: steady-state flow under natural conditions, i.e. no pumping. This scenario was used for inverse calibration of T.

SCENARIO 2: activating the groundwater pumping wells in a steady-state model (scenario 2a) and transient model (scenario 2b). For public water supply a volume of 21.5 $hm^3 yr^{-1}$ (Table 3) was distributed among the screened public supply wells, which coincidentally corresponds to the extracted volume in the hydrological year of 04/05 (though half the volume was withdrawn from the Regional Water Utility wells, see Fig. 1 and Table 2). For irrigation an annual withdrawal of 31 $hm^3$ was considered (Nunes et al., 2006b), distributed over 150 irrigation wells inserted into the model (located within the irrigated areas of Fig. 2). Transient simulations were performed from January 2003 to April 2006, therefore including the drought period of 2004–2005. In this case irrigation wells were only activated from May to September of each year. Trials were performed with different storage coefficients. Many of the runs provided relatively pessimistic scenarios when compared to observed data, simulating the inversion of the hydraulic gradient and consequent seawater intrusion from the Arade estuary. A final storage coefficient of $2.5 \times 10^{-2}$ was used, which provided reliable simulations for most years, except for the final period of the severe and long-lasting drought in 2005.

SCENARIO 3: reducing natural groundwater recharge in the steady-state model by 40%, based on the predicted decrease in rainfall by 2100 (Santos and Miranda, 2006), while maintaining the groundwater withdrawal volumes for public supply and irrigation.

These scenarios should allow the correct identification of the present-time situation and comprise verisimilar scenarios for reduced groundwater availability in the future, either due to a decrease in precipitation or caused by the increase in water consumption by economic uses.

#### 4.1.2 Results and discussion

Regarding scenario 1, the modelled equipotential map in Fig. 6 reveals a steep hydraulic gradient in the NE, where groundwater flow is towards the south. In the western sector the hydraulic gradient is very small and E-W flow direction prevails. The complex flow pattern in the NE is related to the presence of sectors where local karstic structures have a poor hydraulic connection with the main discharge area of the aquifer. Such areas can be characterised as "systematic positive regional anomalies of hydraulic head". Their representation as "low T zones" (Fig. 5) is an assumed simplification of the flow domain that derives from its representation as a single continuum equivalent porous media, which does not interfere with a correct and detailed regional analysis of the hydraulic heads and the water balance. Equivalent porous media models have been successfully applied to karst aquifers by other authors (e.g. Scanlon et al., 2003), the main limitation being that they cannot accurately simulate local scale directions or rates of water flow in the aquifer. If the latter is required, more sophisticated analytical or numerical representations of the flow domain should be used, considering the presence of discrete conduits or fractures (Monteiro and Ribeiro, 2002; Monteiro and Achour, 2005).

Activating the wells in the steady-state model (scenario 2a) results in a lowering of the hydraulic heads in the central and western sector (Fig. 6), where all water-consuming activities are concentrated (see Fig. 2). Despite this fact, mean annual recharge is high enough to avoid regional overexploitation on an average yearly basis, as indicated by the positive aquifer water balance (mean outflow Q of 40.9 $hm^3\,yr^{-1}$). Short-time local overexploitation can occur during longer-lasting dry periods, as is shown for the transient simulation (scenario 2b), depicted in the hydrograph of Fig. 6. This hydrograph shows the transferences between the aquifer system and the Arade estuary. Negative discharges represent outflow from the aquifer and positive values indicate estuarine water entering the aquifer, as result of gradient inversions between the aquifer system and the Arade River. It can be observed that every year outflow is high during the rainy season and drops significantly during the months of May to September, triggered by the pumping activities, particularly those from irrigation wells, which only pump during these four months without recharge. In June 2005, the model simulates a gradient inversion, so that water from the Arade estuary enters the aquifer. Field observations confirm that gradient inversions did occur, however only on a diurnal basis, where the input of saltwater in the aquifer was followed by water output with decreasing salinity until the next rise of the sea tide. Therefore, these transferences were very local as freshwater discharge continued to occur, further confirmed by the absence of any sign of water quality degradation in the aquifer. Despite the need for further adjustment, the model clearly demonstrates the impact of recharge episodes and groundwater extractions on aquifer discharge and the associated risk of seawater intrusion, which can be extrapolated to future scenarios.

Such a future scenario has been simulated for the steady-state model, represented by scenario 3 in Fig. 7, which considers a 40% reduction of natural recharge, with identical pumping activity. Hydraulic heads drop significantly in the entire area, up to 120 m or 60% in the northeast and more than 90% in the west, where the hydraulic gradient is extremely small. There are no signs of seawater intrusion, since the regional mean annual water budget continues to be positive, though it comprises only 6% ($3.5\,m^3\,yr^{-1}$) of natural recharge. In other words, it would be extremely difficult to meet ecological demands for the groundwater dependent ecosystems, whereas the lowering of hydraulic heads would significantly reduce or stop spring discharge. In addition, there would be a need for deeper wells with potentially lower available water volumes due to lower storage capacities. As the steady-state simulation represents an average, long-term situation, evidently the risk of gradient inversion during the dry season or during longer-lasting droughts is largely increased. This is further illustrated by the shaded areas in Fig. 7 that show the evolution of the area with a (steady-state) hydraulic head ($h$) <1 m, along the three scenarios. Whereas from scenario 1 to 2, this area only increases slightly, in scenario 3 the area extends as far as 14 km inward. Here, negative heads and gradient inversions are likely to occur during summer months and almost certain in dry years. The spatial and temporal extent to which these gradient inversions cause seawater intrusion and subsequent water impairment for human consumption or irrigation requires further evaluation with the aid of predictive transient model simulations.

## 5 Conclusions

Groundwater constitutes an important alternative source for an IWRM in the Algarve region, due to its overall high quality and availability. The application of quantitative and qualitative well screening criteria resulted in the selection of six aquifer systems that can be exploited for public water supply. Some have a modest contribution in terms of water volume, but they can be particularly relevant for water treatment, due to their natural characteristics, i.e. high degree of hardness and low turbidity. In total more than $1000\,\mathrm{l\,s^{-1}}$, around 50% of the regional water demand, can be used for the MPWSS, after appropriate disinfection and iron and manganese removal.

The standard violation indices (SVIs) have proven to be useful tools for the spatial monitoring of groundwater quality and potability. Their application is simple and based on drinking water guidelines, so that their interpretation is unbiased. Aggregation also implies a loss of information, and trends of decreasing groundwater quality may be hidden if parametric values have not yet been exceeded. Moreover, the SVIs used in the present study all concern parameters that are simultaneously indicative for groundwater chemistry and drinking water quality, so that the selection criteria are only valid for this water source.

Due to its large area and significant recharge, aquifer system M5, known as Querença-Silves, constitutes the most important groundwater reservoir. Despite its significant exploitation, with 40–50 $\mathrm{hm^3\,yr^{-1}}$ for irrigation and public water supply, no negative consequences have been observed so far, concerning seawater intrusion or ecological degradation. The results obtained by inverse calibration of the regional finite element flow model allowed a significant improvement of the simulation reliability of the observed regional flow pattern, owing to a more accurate characterization of the hydraulic head spatial distribution.

Transient simulations clearly demonstrate the impact of recharge episodes and groundwater extractions on aquifer discharge and the risk of water quality and ecological degradation. The inclusion of oscillatory boundary conditions in the model, incorporating the tidal effect, as well as the spatial analysis of the storage coefficient to reflect its heterogeneity in the aquifer system, are subject to further study, as these particular conditions are related with the understanding of the risks of saltwater intrusion.

The results obtained from predictive modelling point out the need to prepare social and technical tools to alleviate the combined impact from future climate changes and water demand increases in the region. If climate scenarios for effective water recharge at the end of the century are correct, and if no action is taken in the meantime, salinization of groundwater and destruction of ecosystems seem inevitable. Future allocations of water among users will necessarily have to be balanced between socio-economic development and the preservation of natural assets. These are questions that involve inter-generational transference of utilities and require broader definitions of sustainability than those relying essentially on the physical equilibrium of the environment. The present article does not study these matters, but may contribute to helping society, stakeholders and decision makers decide as to the optimal present and future allocation of water resources, based on a more profound understanding of the nature of the problem.

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**Table 1.** Characterization of aquifer systems with regional expression in the Algarve.

| Aquifer system | | Main aquifer lithology | Area ($km^2$) | Mean $Q$[a] ($l\,s^{-1}$) | Recharge ($hm^3\,yr^{-1}$) | Withdrawal 2005 ($hm^3$) |
|---|---|---|---|---|---|---|
| M1 | Covões | lmst[b], dlmt[c] | 22.56 | 15.5 | 6.0 | 0.36 |
| M2 | Almádena – Odeáxere | lmst, dlmt | 63.49 | 5.6 | 16.6 | 0.55 |
| M3 | Mexilhoeira Grande – Portimão | lmst, dlmt, sand | 51.71 | 8.3 | 10.0 | 1.18 |
| M4 | Ferragudo – Albufeira | lmst, sand | 117.1 | 5 | 10.0 | 0.40 |
| M5 | Querença – Silves | lmst, dlmt | 317.85 | 11.1 | 93.4 | 21.44 |
| M6 | Albufeira – Ribeira de Quarteira | lmst, dlmt | 54.55 | 9.4 | 10.0 | 0.55 |
| M7 | Quarteira | lmst, dlmt, sand | 81.19 | 9 | 15.0 | 0.13 |
| M8 | S. Brás de Alportel | lmst, dlmt | 34.42 | 4.2 | 5.5 | 0.34 |
| M9 | Almansil – Medronhal | lmst, dlmt | 23.35 | 7 | 6.5 | 1.92 |
| M10 | S. João da Venda – Quelfes | sand; lmst, marl | 113.31 | 7.0; 5.5 | 9.0 | 0.49 |
| M11 | Chão de Cevada – Qta. João de Ourém | lmst, dlmt | 5.34 | 6 | 2.0 | 0.51 |
| M12 | Campina de Faro | lmst, sand | 86.39 | 6 | 8.3 | 0.13 |
| M13 | Peral – Moncarapacho | lmst | 44.07 | 2.8 | 10.0 | 0 |
| M14 | Malhão | lmst, dlmt | 11.83 | 14.7 | 3.0 | 0.33 |
| M15 | Luz – Tavira | lmst, sand | 27.72 | 5.6 | 4.8 | 0 |
| M16 | S. Bartolomeu | lmst, dlmt | 10.6 | 8.2 | 3.0 | 0 |
| M17 | Monte Gordo | sand | 9.62 | 1.5–3.0 | 3.0 | 0 |

[a] yield, [b] limestone, [c] dolomite



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**Table 2.** Definition of Standard Violation Indexes (SVIs) used in the present study.

| Standard Violation Index | Definition | $t_0 - t_u$ | $NA$[a] | $NV$[b] | $\%_V$ |
|---|---|---|---|---|---|
| $SVI_{Tot}$ | All parameters | 1995–2005 | 38 362 | 3939 | 10.3% |
| $SVI_{Tox}$ | Toxicity parameters | 1995–2005 | 5670 | 15 | 0.3% |
| $SVI_{Microb}$ | Microbiological parameters | 1995–2005 | 3040 | 1077 | 35.4% |
| $SVI_{NO_3,Cl}$ | Nitrate and Chloride | 1995–2005 | 5708 | 1029 | 18.0% |
| $SVI_{Fe,Mn}$ | Iron and Manganese | 1995–2005 | 4339 | 723 | 16.7% |

[a] Number of analyses, [b] Number of violations



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**Table 3.** Relation between available municipal well yields and mean annual recharge.

| Code | Name | $Q^{a}_{scr}$ ($hm^3\,yr^{-1}$) | $Q^{b}_{agr}$ ($hm^3\,yr^{-1}$) | $Q^{c}_{tot}$ (%$R_n$) | $Q^{d}_{scr,Rn}$ ($hm^3\,yr^{-1}$) | $Q^{e}_{scr,0.6Rn}$ ($hm^3\,yr^{-1}$) |
|---|---|---|---|---|---|---|
| M2 | Almádena – Odeáxere | 5.2 | 2.6 | 47% | 5.2 | 5.2 |
| M3 | Mexilhoeira Grande – Portimão | 2.9 | 2.0 | 61% | 2.9 | 2.8 |
| M5 | Querença – Silves | 21.5 | 31.0 | 56% | 21.5 | 21.5 |
| M8 | S. Brás de Alportel | 5.5 | 1.0 | 117% | 4.5 | 2.3 |
| M9 | Almansil – Medronhal | 6.0 | 2.0 | 123% | 4.5 | 1.9 |
| M14 | Malhão | 2.5 | 0.5 | 98% | 2.5 | 1.3 |
| Total | | 43.6 | 42.1 | | 41.1 | 35.0 |

[a] installed yield from selected municipal wells based on screening procedure
[b] groundwater consumption rate of agriculture
[c] $Q_{scr}+Q_{agr}$, as % of mean annual recharge $R_n$
[d] maximum available yield from municipal wells so that $Q_{tot}$ does not exceed 100% of $R_n$
[e] maximum available yield from municipal wells so that $Q_{tot}$ does not exceed 60% of $R_n$



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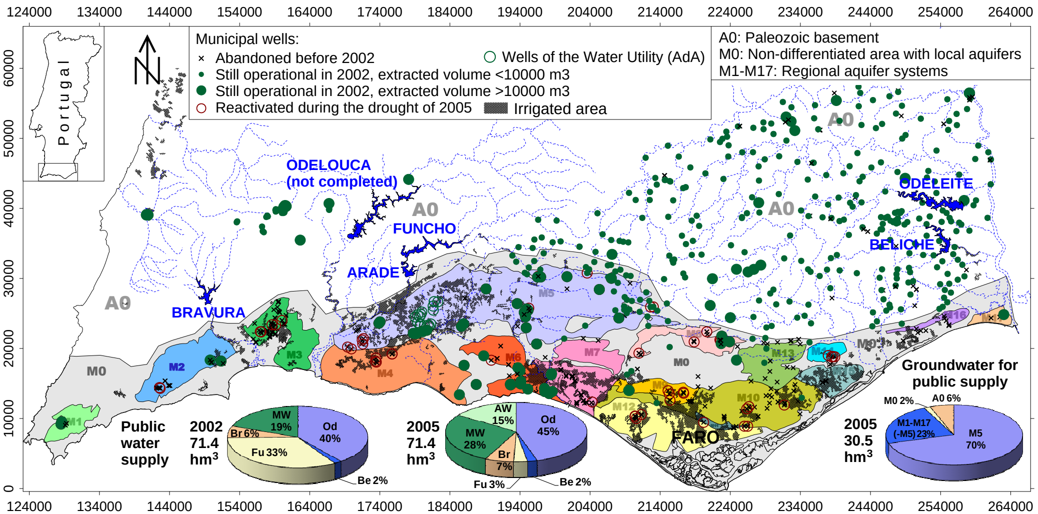


**Fig. 1.** Location of the municipal wells and aquifer systems with regional expression in the Algarve; also shown are the main rivers and surface reservoirs: Od=Odeleite, Be=Beliche, Fu=Funcho, Br=Bravura; groundwater wells: AW=AdA Water Utility, MW=Municipal.



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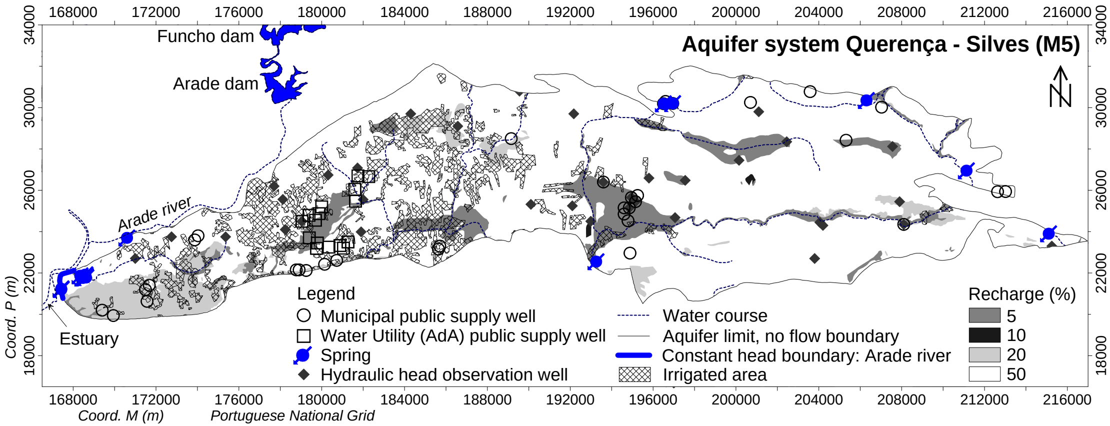


**Fig. 2.** Spatial distribution of recharge expressed as percentage of precipitation and boundary conditions of the Querença-Silves aquifer system. Also indicated are public supply wells, hydraulic head observation wells, springs and irrigated areas.

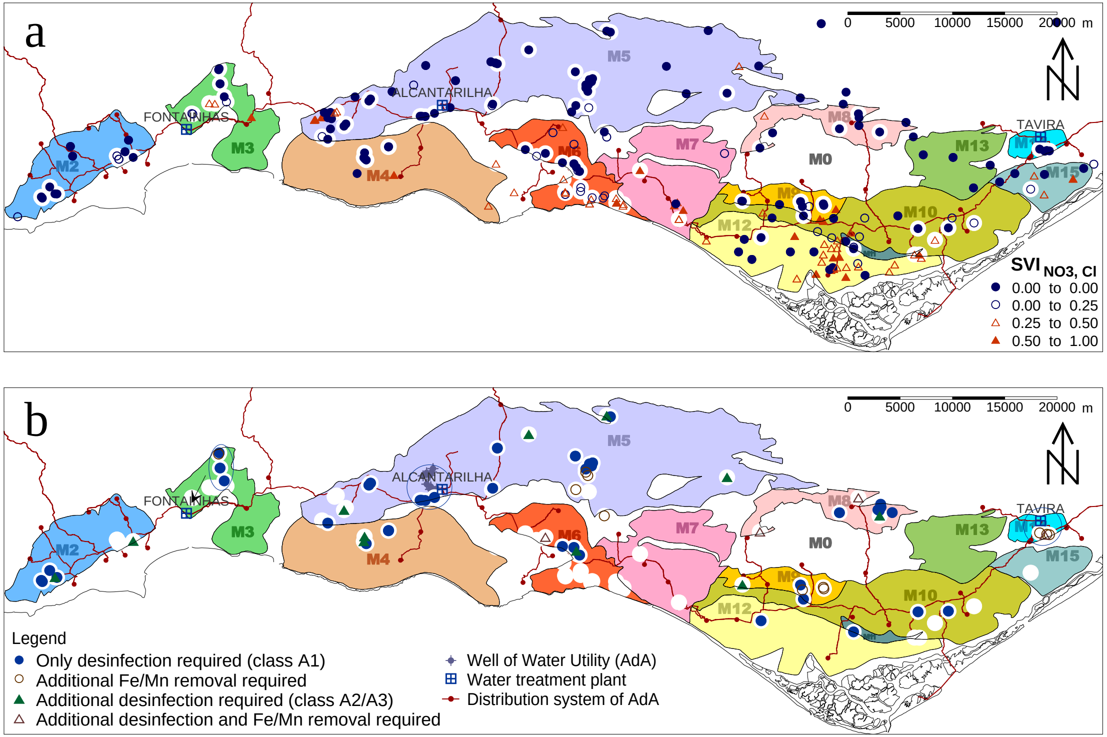


**Fig. 3.** **(a)** (top). Spatial distribution of $SVI_{NO_3,Cl}$ for municipal wells (white background) and private wells in the main aquifer systems of the Algarve; **(b)** (bottom) Location of municipal wells selected by screening procedure, indicating different treatment requirements; no symbols are indicated (only white background) for wells that were excluded during the screening process; also shown is the distribution system of the Water Utility; blue circles indicated wells whose groundwater could be diverted to treatment plants located nearby.



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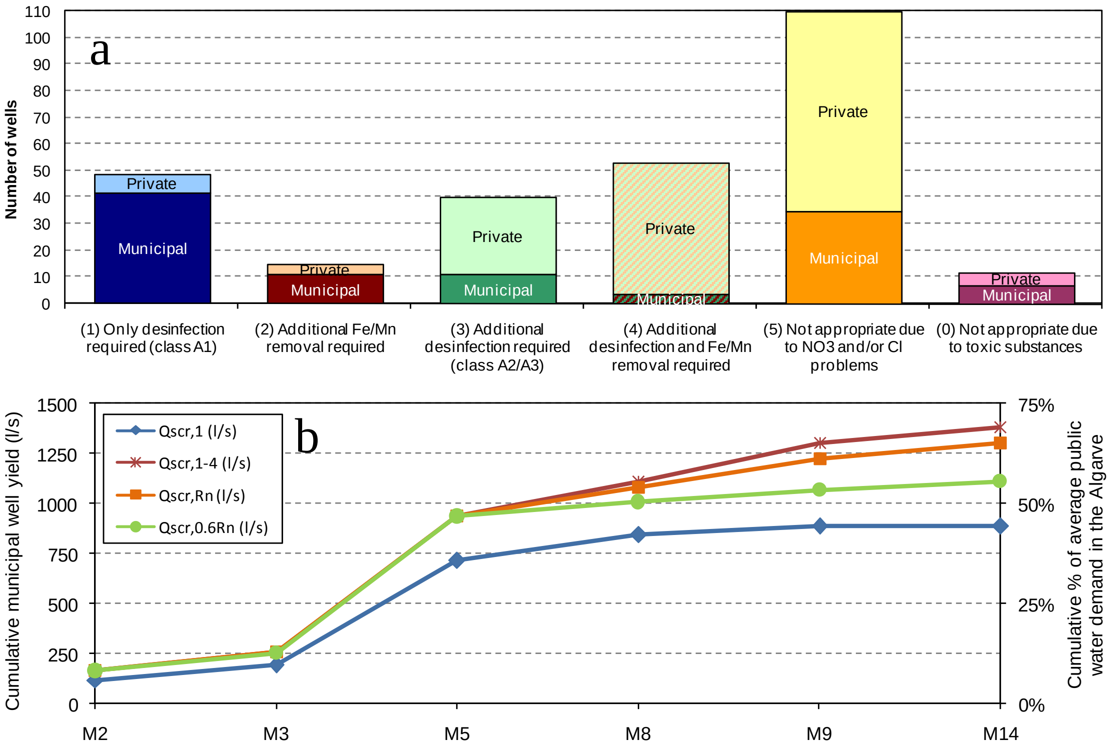
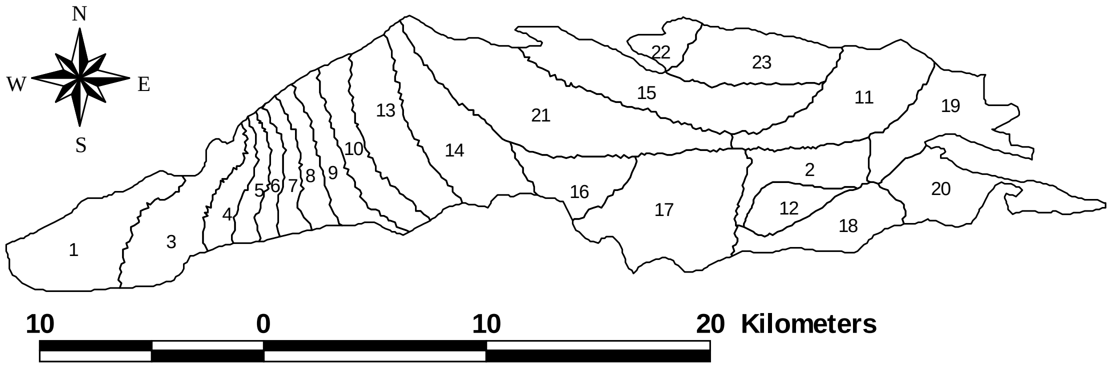



**Fig. 4. (a)** (top) Results of SVI calculations and application of selection criteria for each treatment class; **(b)** (bottom) Absolute and relative values of cumulative well yield for the main aquifer systems; relative values are based on continuous pumping and compared to the average public water demand in the region. Qscr,1=installed yield from selected municipal wells of class 1 of the screening procedure; Qscr,1–4=installed yield from selected municipal wells of class 1–4 of the screening procedure; Qscr,Rn=maximum available yield from municipal wells so that total withdrawals do not exceed 100% of natural recharge (Rn), Qscr,0.6Rn=maximum available yield from municipal wells so that total withdrawals do not exceed 60% of Rn.




| Zone | T ($m^2 s^{-1}$) | Zone | T ($m^2 s^{-1}$) | Zone | T ($m^2 s^{-1}$) |
|---|---|---|---|---|---|
| 1 | 1.35E+00 | 9 | 5.16E-01 | 17 | 2.84E-02 |
| 2 | 2.39E-01 | 10 | 4.78E-01 | 18 | 5.41E-04 |
| 3 | 1.09E+00 | 11 | 1.94E-03 | 19 | 7.50E-03 |
| 4 | 6.65E-01 | 12 | 6.95E-03 | 20 | 7.48E-02 |
| 5 | 5.65E-01 | 13 | 5.09E-01 | 21 | 2.57E-03 |
| 6 | 4.98E-01 | 14 | 6.71E-01 | 22 | 5.00E-04 |
| 7 | 5.10E-01 | 15 | 5.00E-04 | 23 | 5.00E-04 |
| 8 | 5.08E-01 | 16 | 6.86E-01 | | |

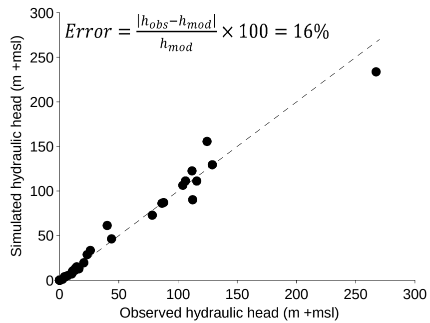



**Fig. 5.** Map with established zonation for estimation of transmissivity (top), optimized values obtained by inverse calibration (bottom left) and resulting plot of modelled versus observed hydraulic heads (bottom right).

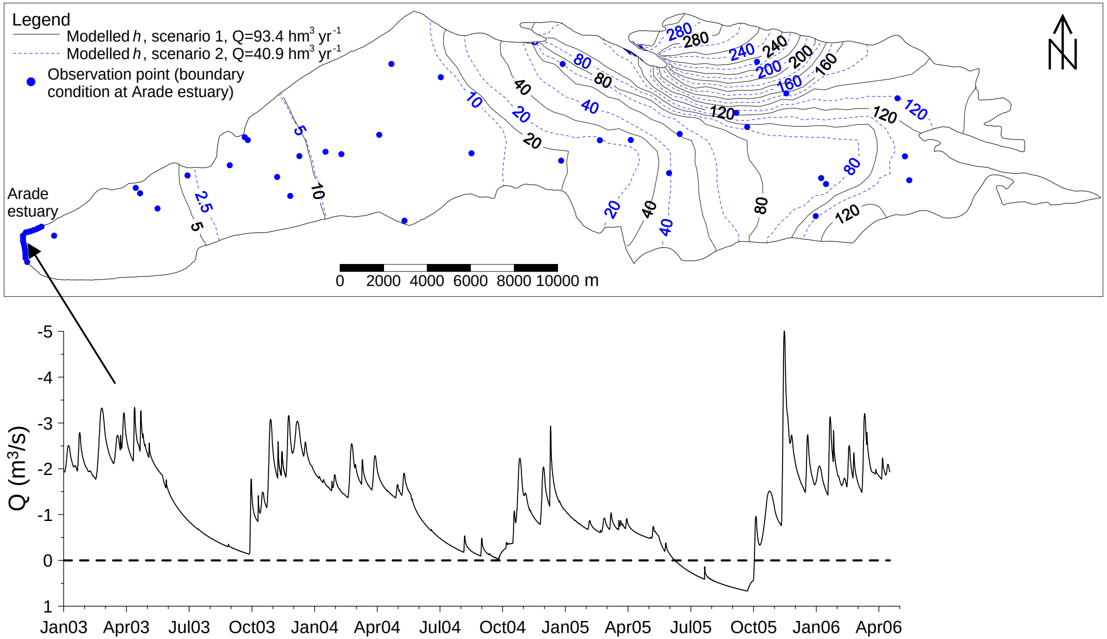



**Fig. 6.** (top) Spatial distribution maps of simulated hydraulic heads under steady state conditions, with and without groundwater pumping (scenarios 1 and 2, respectively); (bottom) hydrograph of simulated transferences between the aquifer system and the Arade estuary using daily values of recharge for a period between 1 January 2003 and 24 April 2006; negative discharges represent outflow from the aquifer.

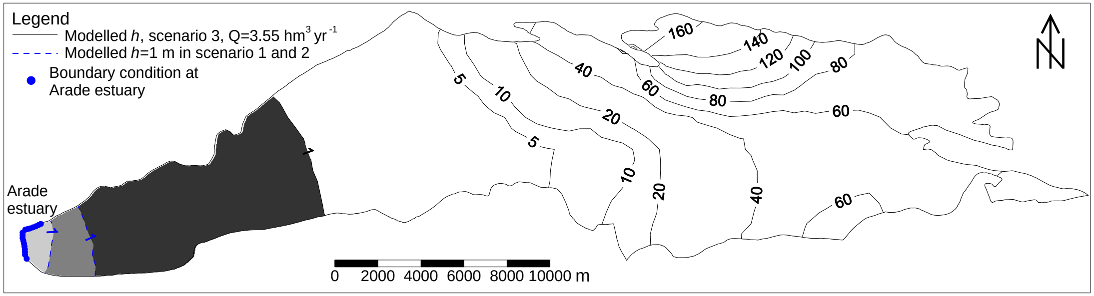



**Fig. 7.** Simulated impact of a 40% reduction of mean annual recharge on spatial distribution of hydraulic heads under steady-state conditions with groundwater abstractions (scenario 3); the shaded areas indicate the increase of the area with hydraulic head ($h$) <1 m, from scenario 1 (light) to 3 (dark).



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