CHAPTER 6

Intensive groundwater development in coastal zones and small islands

A.D. Koussis [(1)] & A. Kotronarou [(2)]
Institute for Environmental Research and Sustainable Development, National Observatory of Athens, Greece
[(1)] *akoussis@env.meteo.noa.gr*
[(2)] *natasha@env.meteo.noa.gr*

G. Destouni [(3)] & C. Prieto [(4)]
Royal Institute of Technology, Stockholm, Sweden
[(3)] *gia@kth.se*
[(4)] *carmen@aom.kth.se*

ABSTRACT: By examples from around the world, we show the vulnerability of coastal aquifers, emphasising sea intrusion from intensive groundwater development. Given a continuing high demand pressure on coastal aquifers and the high risks and costs of pollution, the issue is to make coastal zone development sustainable. This requires consideration of alternate water sources and associated prediction uncertainties, e.g. due to climate change, subsurface heterogeneity and demand/demographic evolution. We outline a research methodology and an integrated tool for investigating the feasibility of an aquifer management strategy that uses desalination of brackish groundwater, coupled with control of sea intrusion via recharge of treated wastewater. Through the case study of an aquifer in the island of Rhodes, we show that this can be an economically viable, sustainable aquifer management scheme in seasonally water-stressed, semi-arid coastal zones and islands.

1 SUSTAINABLE DEVELOPMENT AND MANAGEMENT OF WATER RESOURCES

Society owes the increased standard of living, to a large extent, to technological advances in the various production processes. However, as resources are utilised in the production of goods, unwanted by-products result too (the term goods includes material goods as well as energy; similarly, by-products can be material wastes or waste energy). Today, society has reached consensus that wastes should not be simply disposed of in the cheapest possible way, realising the negative environmental implications of such practice. Instead, the entire production scheme of goods should be managed economically as well as from an environmental point of view. The OECD countries have embraced the concept of ecologically sustainable development, eyeing economic growth and environmental protection as complementary goals. Attaining these goals presumes effective management of natural resources and maintenance of sustained yields from ecosystems (OECD 1993).

The NRC (1991) publication, *Opportunities in the Hydrologic Sciences,* highlights the essential role of water for human life. Its unique physical and chemical properties enable water to be *elixir of life, climatic thermostat* and *global heat exchanger*. Elixir because, as an almost universal solvent, it ensures nourishment of cells and removal of their wastes, thermostat since its high specific heat, and correspondingly large thermal inertia, make it the flywheel of the global heat engine, and heat exchanger due to its high latent heat value. Furthermore, water is essential for agricultural food production, and thus a foundation for the prosperity of mankind, which explains agriculture as the locomotive for the development of water resources (irrigation accounts for ~75% of current world water consumption).

The management of resources aims at their optimal disposition in space and time, with control of side effects. The increased pressures on water resources result from a growing world population and from its legitimate expectations of a higher standard of living, especially in the developing countries. It is estimated that, over the past three centuries, water withdrawals have increased by a factor of 35 while the world population has increased 8-fold. But exploitation of a finite resource is limited. In contrast, sound management promotes sustainable development, ensuring that current use of a resource does not compromise its use by future generations (World Commission on Environment and Development 1987). For this to succeed, authorities should educate the public to appreciate water scarcity; in particular accepting re-cycled treated wastewater as a new source of water. Furthermore, conservation and proper use of water should be promoted also through water pricing (*the user pays*) and through penalties for degradation of water quality (*the polluter pays*).

The penetrating review of Sophocleous (1998, and also this volume) of the much-debated *safe yield* concept underscores the complexity of sustainable development in an era of a maturing water economy. This era is characterised by increasing competition for access to fixed resources, a growing risk for water pollution and sharply higher economic, social and environmental costs of development. Sophocleous concludes that, to turn the principles of sustainable development into achievable policies, solutions must be based on fundamentally sound hydrologic analyses and technologies. Yet application of this self-evident thesis is a formidable task. For example, solutions should acknowledge the links between surface and groundwater or water quantity and quality. Many of these ideas are acknowledged, or even stated as explicit goals in official documents such as the European Union's Water Framework Directive. However their translation into practice requires the political will to confront an array of interest groups.

2 MANAGEMENT OF GROUNDWATER IN COASTAL ZONES AND IN ISLANDS

2.1 *Considering climate change and uncertainty*

In an era of competition for limited water resources, their management must consider the impacts of climate change, e.g. sea level rise. Historic patterns of the hydrologic cycle and the statistics that, presumably, describe stable behaviour (*stationarity*) may no longer be taken for granted, and impacts at the local scale are especially hard to predict. Current knowledge of climate change may be unable to give a definite answer regarding e.g. future annual precipitation values (increase or decrease), but variability will certainly increase, making extremes more prominent. Therefore management of water resources systems should be adaptive and should quantify uncertainty.

Uncertainty, and thus risk, is implicit in the use of probabilities, a time-honoured tool of engineering practise; it can be quantified by statistical measures such as the mean and the standard deviation. Uncertainty derives from the stochastic nature of climatic variables and from the heterogeneity of the subsurface that is relatively inaccessible to detailed measurements. Yet demographics may be a greater source of uncertainty. We can obtain stochastic outputs and their statistics by using stochastic models, or by driving deterministic models with random inputs.

A stochastic methodology has the advantage of yielding answers *and* estimates of their uncertainty, thus allowing to assess the reliability or, its complement, risk of solutions. The deterministic solution coincides with the *expected* solution in a stochastic framework only if there are no products of stochastic variables in that solution. After ensemble averaging (over all realisations), such products contribute co-variances that do not vanish necessarily. As a result, in the stochastic solution appear the deterministic solution terms and additional terms involving variances and co-variances of stochastic variables. Thus it is not certain that the deterministic estimate is always the most conservative. Recalling the analogy to the elementary treatment of turbulence can help readers familiar with that subject. After velocity decomposition, in a mean field and a fluctuation around it, and time averaging of the Navier-Stokes equations, the inertia terms contain also products of fluctuations. Correlated fluctuations give rise to turbulent shear stresses that increase flow resistance.

2.2 *Intensive development of coastal aquifers*

Coastal zones are often characterised by high population densities and intense economic activities. For example, within a distance of 50 km from the 89,000 km-long coastline of the European Union lives about half of its population of ~380 million persons (EC 2001). Agriculture and tourism are often prominent activities, placing heavy seasonal demands on the groundwater resources. Of course, the climatic, geologic, soil and hydrologic conditions that prevail, e.g. at the Mediterranean and at the Atlantic or the Baltic coasts, differentiate the groundwater problems. The water resources of islands, especially small and remote islands, are under additional stresses due to their geographic isolation that precludes such solutions as inter-basin water transfer, except via expensive transport by tank ships.

Two significant and common threats to the groundwater resources of coastal zones and islands are sea intrusion and infiltration of pollutants on the landside. Seawater intrusion arises from the intensive development of groundwater itself and causes a large-scale, largely human-induced aquifer contamination by a natural chemical. Mixing freshwater with 2% of seawater (salinity ~35,000 ppm TDS, Total Dissolved Solids) raises water salinity sufficiently to make it unsuitable for drinking (potable water standard is 500 ppm TDS), 5% mixing makes freshwater unsuitable for irrigation (Custodio & Bruggeman 1987), excluding specially salt-resistant plants. Aquifer vulnerability to contaminants from the landside comes from a variety of sources and activities. For example, open land development, with reliance on septic tanks, can lead to elevated nitrate concentrations. The same, and more strongly, holds for agriculture with intense fertiliser use, often adding pesticides as a problem. The common proximity of coastal aquifers to the surface increases their vulnerability to pollution; e.g. in the Bahamas 90% of fresh groundwater lenses are within 1.5 m from the surface.

The issues raised here are elaborated further in the remainder of the chapter. Section 3 outlines several cases of intensive groundwater development in coastal aquifers, giving a broad perspective. Section 4 presents a management concept for coastal aquifers, with re-use of treated wastewater and desalting of brackish groundwater. Application of this concept to an aquifer in Rhodes is demonstrated in Section 5. The chapter closes with the section on conclusions.

3 CASES OF INTENSIVE GROUNDWATER DEVELOPMENT IN COASTAL AQUIFERS

3.1 *Example cases*

- The Akrotiri Aquifer, Cyprus (Koussis 2001).

The aquifer (~40 km^2) is located in the Akrotiri peninsula on the south coast of Cyprus, is largely unconfined, consisting mainly of river deposits (10 to 100 m thickness, n ~0.2, $T = 1{,}000$–$2{,}500$ m^2/d), and borders a Salt Lake, Figure 1. The Akrotiri aquifer is exploited heavily (~10 Mm^3/yr, ~300 wells) for the water supply of Limassol, the British bases in the area and several smaller communities, and for agriculture.

Challenges: Declines of hydraulic head and sea intrusion have been documented. Several coastal wells on the western side have been abandoned and pumping on the eastern side was curtailed by ~25% after 1997.

Solution approach: To limit seawater intrusion, the aquifer is recharged artificially with water from reservoirs located outside the basin applied on spreading grounds over the aquifer. Recharge was reduced during the relatively dry decade of the 1990s and pumping was reduced. Irrigation return flows also replenish the aquifer. Surface spreading of treated effluents of the Limassol wastewater treatment plant is scheduled to enhance recharge. In addition, the Water Development Department of Cyprus plans to augment the water supply with desalinated seawater.

- Israel and Gaza Strip (Melloul & Zeitoun 1999)

The Israeli and Gaza Strip Coastal Aquifer consists of Pleistocene age sandstone, calcareous sandstone, silt, and intercalated clays and loam that can appear as lenses. Marine clays and shales of the Neogene age form the aquifer base. The base is sloping at ~1% towards the sea, where the aquifer thickness reaches about 160 m, as shown in Figure 2. Over the first ~5 km from the sea, clay lenses divide the aquifer in three major sub-aquifers (A, B, C) that form separate hydrologic units. The coastal aquifer is used for long-term water storage and as source of drinking and irrigation water that is extracted by ~3,000 wells.

Challenges: The recommended annual withdrawal from the Coastal Aquifer is around 280 Mm^3, in recent years however annual pumping ranged from 350 to 400 Mm^3. Widespread seawater intrusion has occurred. Especially affected are the upper sub-aquifers A and B, where the abstractions from shallow wells are more economical and significant; chloride concentrations exceed 1,000 mg/L and are increasing at over 10 mg/L/yr. In wells along a 100 km north-south front (down to the Gaza Strip, Fig. 3) that were located from 500 to 1,500 m from the seashore, salinity increased rapidly in the 1980s, after a long period of modest variation. In one to three years, chloride values increased at a rate of 300 mg/L/yr, reaching in the Gaza Strip nearly 2,500 mg/L.

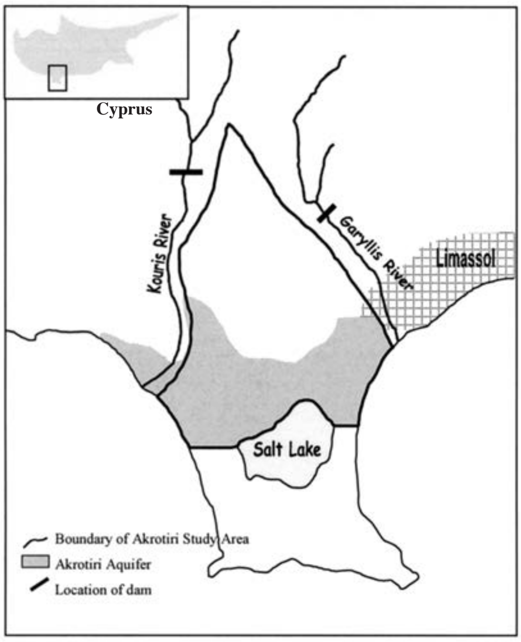

Figure 1. Map of Akrotiri Aquifer, Cyprus.

Solution approach: Israel has integrated the re-use of wastewater in a national water management plan, re-cycling ~70% of its wastewater. The National Water Carrier system uses two pipelines, one distributing freshwater, the other treated wastewater. Since the 1960s, biologically treated wastewater from the Tel Aviv metropolitan area is used to recharge the coastal aquifer in the Dan region (~90 Mm^3/yr), counteracting the overdraft. The soil is considered a reactor and recharged water is extracted only after a certain residence time. Recent management decisions promote use of groundwater for drinking purposes, rather than for irrigation, as well as measures to reduce evaporation losses. A monitoring network has been implemented to track salinity changes in the Coastal Aquifer, which has been divided in squares formed by a set of columns (parallel to the coastline) and strips, as shown in Figure 3.

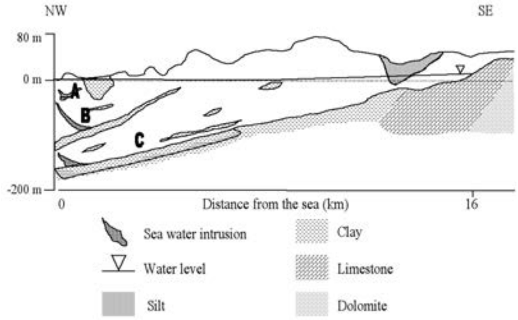

Figure 2. Hydrogeologic cross-section of the Coastal Aquifer of Israel and Gaza Strip (adapted from Melloul & Zeitoun 1999).

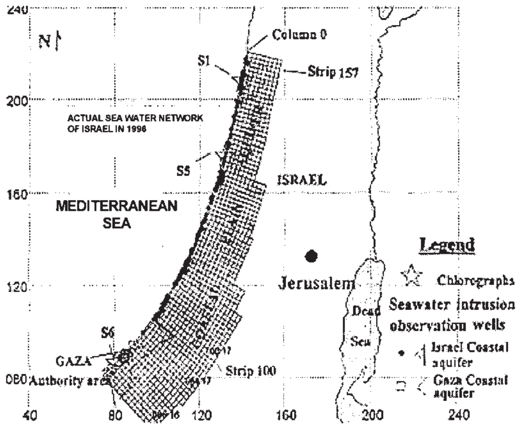

Figure 3. Map of seawater intrusion network (adapted from Melloul & Zeitoun 1999).

- Netherlands (Stakelbeek 1999).

The coastal aquifer under the polder region, behind the North Sea coast dune area, is used as source of potable water. This sand aquifer consists of four units that are separated by clay and loam layers (Fig. 4). In the lowest unit the groundwater is brackish.

Challenges: To avoid salinisation of the aquifer, abstractions have been reduced in the last 30 years. Recharge and subsequent abstraction of pre-treated surface water must compensate the reduction, using the water passage through soil as method of disinfection. Deep-well injection is preferred due to its low space demands and mild environmental impacts. A rise in sea level of 0.6 m in 100 years is expected to accelerate salinisation significantly (Oude Essink 1999).

Solution approach: Since 1990, a deep-well infiltration system of 5 Mm^3/yr capacity is in operation. It consists of 20 infiltration and 12 production wells situated in the aquifer unit between 50 and 100 m deep; the aquifer is separated from the brackish unit by a thin clay layer. To prevent upconing (see Appendix) of brackish water near the overlying abstraction wells, the wells are arranged in a rhombus-like pattern, with the infiltration wells in the middle and the abstraction wells on the well field sides, and a 10% over-infiltration is applied. All the over-infiltrated water cannot be recovered, but it enables continuing the abstraction without infiltration for a certain time.

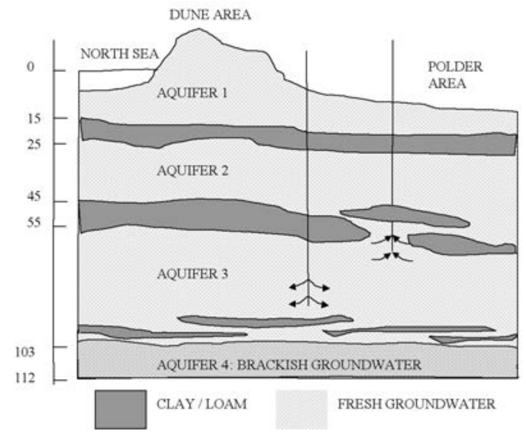

Figure 4. Hydrogeologic section near the deep-well infiltration plant (adapted from Stakelbeek 1999).

• California, USA (Konikow & Reilly 1999).

Orange County is a region south of Los Angeles with semi-arid Mediterranean-type climate (mean annual rainfall 380 mm). The coastal aquifer system is made up of several aquifers, each one consisting of a permeable sand and gravel layer confined between clay and silt layers. A fault zone forms a low permeability barrier. In Orange County, this barrier is interrupted by gaps filled by permeable alluvial deposits. The aquifer is used primarily for the public water supply of the populous Los Angeles metropolitan area.

Challenges: Several incidents of seawater intrusion occurred as early as the mid- to the late-1940s, when massive withdrawals during a long drought caused aquifer levels to drop 5 m.b.s.l. (Fig. 5). Artificial recharge with imported water was applied (assessing those pumping a fee), but in 1956 intrusion progressed up to 5.6 km, as aquifer levels dropped to 7 m.b.s.l. A brief period of recovery in the mid-1960s was achieved through increased recharging, water imports and reduced withdrawals, but it became clear in the late 1960s that sea intrusion continued. A later adopted conjunctive-use policy proved also insufficient to stop sea intrusion.

Solution approach: After seawater had migrated 8 km along a buried channel and several wells were abandoned, it was decided in 1978 to attempt to stop intrusion by creating a hydraulic barrier through injection of treated wastewater. Presently the aquifer is recharged with a mix of wastewater treated to tertiary level (~75,000 m^3/d) and fresh deep-well water (~38,000 m^3/d). Scavenger wells, located seaward of the injection wells, remove brackish water; after desalting, that water is being considered as a potential new source. The system is monitored extensively.

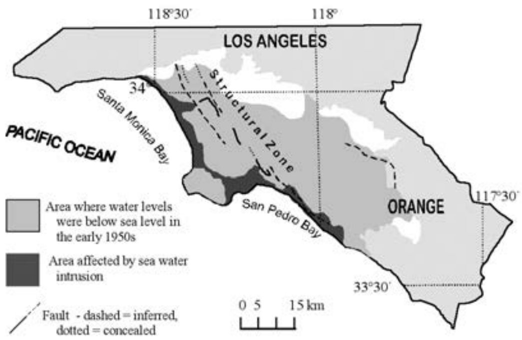

Figure 5. Map of Los Angeles - Orange County Coastal Plain Basin (adapted from Konikow & Reilly 1999).

• Florida, USA (NRC 1993, Konikow & Reilly 1999).

Population 17 million (2000), ~40 million tourists annually. Groundwater supplies 95% of the population with drinking water at a rate of ~5.7 Mm^3/d. The needs of agriculture are met by pumping another 11.5 Mm^3/d. Much of the state is underlain by the highly productive Floridan aquifer (Fig. 6), a largely limestone and dolomite aquifer, in confined and unconfined conditions. The most intensively exploited aquifer is the Biscayne, a shallow, unconfined, limestone aquifer in Southeast Florida. Layers of sand, clay, marl, limestone or dolomite of considerably variable thickness overlie these aquifers.

Challenges: The primary problem is seawater intrusion that is caused by over-pumping. This problem is acute in the Biscayne aquifer, where uncontrolled drainage by canals (from 1909 through the 1930s) lowered water levels by ~2 m in the Everglades. Also major contamination sources are pesticides and fertilisers (2 million tons/yr), ~2 million septic tanks, over 20,000 wells for disposing of storm water, treated wastewater and cooling water, ~6,000 surface ponds and phosphate mines that disturb 3,000 ha each year.

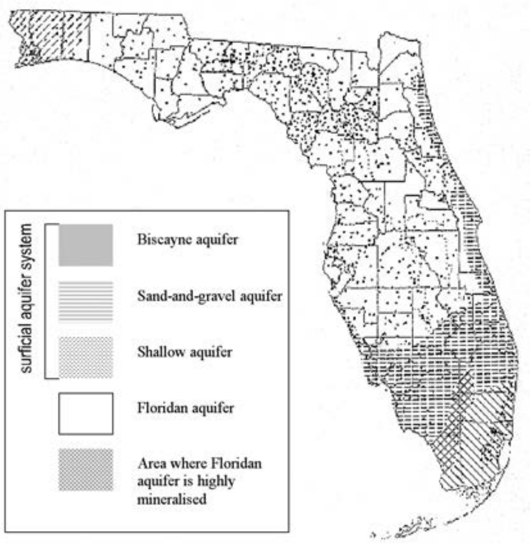

Figure 6. Principal aquifers in Florida and network of sample wells (1990 status) (NRC 1993, reprinted with permission).

Solution approach: Recent emphasis has shifted from enforcement toward a technically based, quantifiable, planned resource protection approach. Control of sea intrusion in the Biscayne aquifer relies on control of water level in the canals. The balance between withdrawals and natural or artificial recharge is to be observed. Contamination is to be prevented; however it is not required that non-degradation standards be met everywhere at all times. A monitoring network has been established.

• The Hawaiian Islands, USA (NRC 1993).
Over 90% of Hawaii's over-one-million inhabitants and the numerous tourists rely on groundwater for their supply of drinking water. Approximately 80% of the population reside in Oahu. The Hawaiian Islands are formed from shield volcanoes composed mainly of very permeable, thin basaltic lava flows. Alluvial and marine origin coastal-plain sediments cover the margins of those volcanic mountains. Groundwater occurs as basal water floating over seawater, as high-level water bodies impounded in areas bounded by natural dikes that intrude the lava flows and as perched aquifers, as shown in Figure 7.

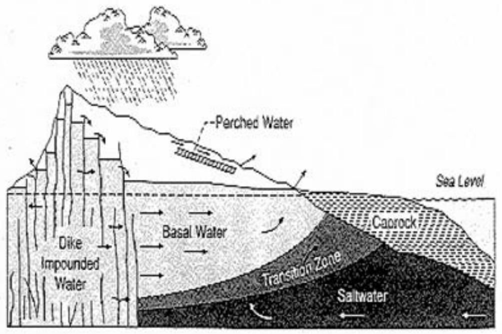

Figure 7. Cross-section of a typical volcanic dome showing the occurrence of groundwater in Hawaii (After Peterson 1972. Reprinted, by permission from Water Well Journal Publishing Company 1972).

Challenges: By far the primary problem is seawater intrusion in coastal areas that is caused by intensive groundwater development, especially on Oahu. Groundwater contamination from non-point sources of agro-chemicals is an increasingly major concern, especially pesticides used for nematode control in the large pineapple monoculture. Several incidents of groundwater contamination from the nematocides DBCP and EDB have been documented since 1977 and 10 wells were closed in 1983. However contamination dates, probably, much earlier, as it is also related to the production of the fumigant DD.

Solution approach: Following the public demand, regulators have adopted a strict zero-tolerance policy on groundwater contamination. Nevertheless, the contamination caused by past and present activities, especially due to pesticide applications, will persist for some time. Seawater intrusion due to over-pumping is to be addressed through better management. Pesticide regulations are being developed, based also on monitoring results. Water from municipal wells at central Oahu, where DBCP, EDB and/or TCP were detected, is filtered through activated carbon, but the treatment cost is passed onto the consumers, not onto the chemicals' producers.

• Barbados (Burke & Moench 2000).
The public water supply of this Caribbean island, of 260,000 inhabitants, which also serves tourism, the island's prime industry, relies almost entirely on a phreatic karst aquifer. The rapid flow in the aquifer makes it vulnerable to pollution and saline intrusion.

Challenges: Protecting the public water supplies is a priority, given the cost and technical difficulty of alternate sources. Agricultural changes from sugar cane to cash crops, urban expansion to pristine aquifer areas, on-site sanitation and industries pose threats.

Solution approach: Strict enforcement of legislation mandating the development of hierarchically controlled zones around public supply sources, based on pollutant travel times, and seawater intrusion prevention through control of the abstraction regimes.

• Greece.
According to an unpublished map of the Hellenic Institute of Geology and Mineral Exploration, sea intrusion has occurred in many parts of the coast of the mainland, mostly along the east side, and especially along the coasts of many Aegean islands, where uncontrolled well-drilling and aquifer exploitation have taken place. The best known case of seawater intrusion caused by over-pumping in Greece concerns the Argolis plain (East Peloponnese), where extensive orange groves are cultivated. The impact of seawater intrusion on some orchards has turned some farmers to the cultivation of salt-resistant artichoke plants. A successful artificial recharge programme, with upland spring water channelled to about 100 wells, was begun in 1994, to counteract saltwater encroachment in an area of ~40 ha. Contamination by fertiliser-derived nitrates is the main problem further upland.

3.2 *A special case: management of groundwater in coastal aquifers in semi-arid climate*

We focus briefly on coastal areas and islands that are located in semi-arid zones, classified according to the UNESCO moisture index, I_h. This is a composite climatic parameter defined as the ratio of the average annual values of precipitation, *P;* over potential evapotranspiration, *PET*.

$$I_h = P/PET \qquad (1)$$

Semi-arid regions are characterised by $I_h \leq 0.5$, which indicates that potential atmospheric losses far exceed precipitation; e.g. in Athens, Greece, $PET > 3P$ (Koutsoyiannis & Baloutsos 2000). In such climatic zones, precipitation varies greatly over the year and extended periods of no runoff and no recharge exist, as e.g. in Southern California. In the European Union, semi-arid regions are Southeast Greece, Southeast Italy plus Sicily, Southeast Spain, parts of Central Spain, the Canary and Balearic Islands, and South Portugal. Other Mediterranean semi-arid regions are Cyprus, Malta, the Middle East and North Africa.

Tourism and agriculture are important activities in these areas, both of which place great stress, at an increasing trend (EEA 1995), on water supply during the dry season. Mild temperatures and good quality soils of the Mediterranean region have led to intensive irrigated agriculture (also increasing trend), with greatest water requirements for crops in the dry period May-September, with only ~10% of mean annual precipitation (López Martos 1998). Tourists visit Mediterranean locales preferentially during the same period and consume about twice the water of an average (local) consumer (López Martos 1998).

On the annual cycle of water demand in semi-arid regions is superposed the cycle of droughts, with an apparently increasing frequency relative to historic patterns. The drought of the first half of the 1990s in Southern Europe showed the region's vulnerability to water shortages. For example, in major cities of Andalusia, Spain, such as Córdoba, Málaga and Sevilla, water supply was cut daily for several hours and the salinity of tap water forced the public to drink bottled water. In the early 1990s, as much as 0.5 Mm^3 of groundwater was pumped daily, mainly from aquifers in Boiotia District (just north of Attica where Athens is located), to augment the supply of Athens. The intensive exploitation disturbed the hydrologic regime of the aquifers around Lakes Yliki and Paralimni, causing a 5–10 m drop of hydraulic head and salinity increases to 900–2,000 ppm (Cl^-) locally. Already since 1984 Athens has been drawing upon the water resources of River Mornos. Since 2001, Mornos Reservoir, located ~160 km from Athens, is receiving the water of River Evinos, the national budget paying again for the supply of Athens.

3.3 *An assessment of the challenges posed by sea intrusion in intensively developed coastal aquifers and a discussion of potential solutions*

The examples sketched in Section 3.1 identified seawater intrusion as a major, and likely the dominant threat to intensively exploited coastal aquifers. Similar aquifer problems, encountered e.g. in Gujurat (India), Indonesia, Malaysia, Thailand or Yemen, were not outlined due to scarcity of specific, readily accessible data. The threat of seawater intrusion is also more acute in semi-arid regions where overdrafts are more likely to be used as means of handling periodic water shortages. An obvious remedy to sea intrusion is to reduce abstractions, but at a loss of needed water resources, which must be made up from other sources. Increasing the system's hydraulic defences can strengthen this measure further, as done in Florida, USA, by maintaining a minimum water level in the system of coastal canals. In California, Cyprus, Israel and the Netherlands, artificial recharge is used to create a hydraulic barrier to the advancement of the saltwater front. In the Netherlands the recharge is from treated water from surface sources, in Israel it is treated wastewater and in California and in Cyprus it is a mix of freshwater and treated wastewater.

Generally, in situations of severe water demand stress, with precipitation-derived resources fixed, alternate sources of water must be considered. The related scientific-technical, economic, legal and societal facets of non-traditional practises must be studied in this context. Two non-traditional sources, with cost-effective potential, are readily identified (World Bank 1995): a) saltwater, which can be treated to potable quality standard, and b) treated wastewater, which can be re-cycled to meet various demands. Re-use of wastewater is limited presently, but can be applied more widely when the public's perception improves. Desalination will be a viable alternative, when its economics are favourable; we show below that this is the case with brackish water.

In locales where the natural resources of coastal aquifers have been either exhausted or are nearing the state of overdraft, public water supply authorities have considered seawater desalination as the solution of choice. For example, desalination of seawater is practised extensively in the Canary Islands (over 90% of the desalination capacity installed in Spain), is also employed in a few islands in the Aegean Sea, e.g. Mykonos and Santorini, and is being considered by the Water Development Department of Cyprus and by the Palestinian Authority. However, the economics of desalting brackish aquifer water are notably more favourable and this circumstance opens up an opportunity for a non-traditional management scheme of the resources in coastal aquifers. As already noted in Section 3.1, desalting of brackish water is also being considered in California.

Re-use of treated wastewater is practised in various parts of the world, but under non-uniform standards. For example, European standards do not exist; as a result different regions use their own rules (Salgot & Pascual 1996). The two most widely used standards are those of the World Health Organisation (WHO) and of the State of California, USA (Title 22). The less stringent WHO standard can be met with simpler technologies; the intent is to replace the use of raw wastewater for irrigation in regions with inadequate technologies of water supply and sanitation and to thus inhibit the spread of waterborne diseases. The California standard aims to ensure that re-claimed water is nearly free of pathogen levels, to be accepted by a health-conscious public [number of USA-regulated drinking water contaminants: 25 in 1980, over 150 in 2000 (WEF & AWWA 1998)].

A sustainable scheme of water resources management in regions where water is scarce can be developed on the premise that intensive exploitation of coastal aquifers inevitably causes a certain degree of seawater intrusion (as not all water is returned as treated wastewater and extractions periodically exceed total recharge). The scheme should consider desalination of brackish waters [waters of slight, 1,000–3,000 ppm TDS, to moderate salinity, 3,000–10,000 ppm TDS (McCutcheon *et al.* 1993)]. The energy required for desalting brackish water is low compared to seawater desalination (Georgopoulou *et al.* 2001) and this can be important in isolated territories, especially small islands not connected to the power grid. The hydrologic budget can be then enhanced by recharging coastal aquifers with treated wastewater, instead of discharging it to the sea. Economics and environmental conditions must be evaluated too.

The development of a tool for assessing various such aquifer management options objectively was the goal of the project entitled *Utilisation of Groundwater Desalination and Wastewater Reuse in the Water Supply of Seasonally Stressed Regions* (acronym *WASSER*). The methodology of the project is outlined in Section 4; a specific application is summarised in Section 5. The project's Final Report (Koussis 2001) gives a more complete account.

4 UTILISATION OF COASTAL AQUIFERS: CONTROL OF SEA-INTRUSION THROUGH RECHARGE OF TREATED WASTEWATER AND DESALINATION OF BRACKISH GROUNDWATER

4.1 *The WASSER concept and its components*

The WASSER concept adds to the practise of controlling seawater intrusion, by recharging a coastal aquifer with treated wastewater, the extraction of fresh and/or brackish groundwater to meet demand, including indirect potable water re-use. Alternative schemes are evaluated objectively through modelling that encompasses the physics, the engineering and the economics of the system. The system consists of natural and of engineered components. The natural subsystem includes the watershed and the underlying aquifer, which are interconnected via the recharge. The engineered subsystem concerns the facilities for desalination, for wastewater treatment and for pumping and storage. A Decision Aid Tool (DAT) was developed to facilitate this work.

Within the shell of DAT are contained an optimisation package, for screening of alternative recharge/extraction scenarios, and a package for the detailed evaluation of scenarios that the user wants to study in depth, based on economic and environmental aspects. The database of DAT contains information on water desalination and wastewater treatment and accepts user-supplied data on water demand and re-use of wastewater. DAT also accepts data obtained by running off-line appropriate models describing the natural system dynamics, for a number of scenarios. Off-line runs are necessitated by the numerical integration of complex models in an optimisation framework, as well as by the CPU-intensive stochastic simulations. The application of the WASSER concept is based on a planning horizon of 20 years, which can be changed to accommodate user needs.

4.1.1 *The natural system*

Starting point is the watershed's surface hydrologic balance. The recharge of the groundwater system derives from measured or estimated climatic inputs (precipitation and climatic parameters controlling evapotranspiration, *ET*) and from irrigation water or other artificial recharge. The aquifer recharge was computed from the hydrologic balance as the *loss* of the surface and the soil to the deep subsurface, i.e.:

$$rech. = [surf.\ input] - [Qsurface + ET + \Delta(soil\ moisture)] \tag{2}$$

The primary hydrologic modelling identified the characteristics of the inputs and outputs of the system, which were used subsequently to develop synthetic time series, with the proper statistics.

The groundwater system dynamics was analysed deterministically and stochastically with the coupled water flow and salt transport model SUTRA (Voss 1984). The stochastic analysis was based on Monte Carlo simulations and considered uncertainty in the estimation of spatial quantities (distribution of hydraulic conductivity field, *K*, i.e. random aquifer heterogeneity) and of temporal processes (randomness of recharge and boundary inflows). The stochastic investigation aimed to quantify uncertainty in the estimation of the salinity field and, through it, in management decisions, specifically in relation to the ramifications of recharging the aquifer with treated wastewater. Management decisions are affected, e.g. by the location of the salt-fresh water interface (e.g. defined by the contour line S = 17,500 ppm TDS), or of the

500 ppm TDS contour line (the potable water standard), or by the salinity of the pumped water.

Care was taken to make the deterministic and stochastic approaches as compatible as possible. The conceptual profile models (2-D in the vertical plane), initial conditions and other field parameters such as the specific yield (effective porosity), except the hydraulic conductivity, were kept the same. Furthermore, the geometric mean of the random K field was chosen to be equal to the constant K-value used in the deterministic simulations. In addition, the total freshwater volumes of natural recharge and boundary inflows in the deterministic and in the stochastic simulations were kept nearly the same. However, despite the almost identity of the *mean hydraulics* in the two approaches, the deterministic and the mean stochastic salinity responses are different, at times significantly so. As expected, the departure from the deterministic behaviour increases when spatial and temporal randomness are considered. In two of the three project case studies, spatial heterogeneity turned out to be a more important source of uncertainty than temporal randomness, whereas in the third both sources were important. The site-specific reasons for such differing results are explained in Section 5.4, in connection with the more detailed discussion on the groundwater dynamics modelling.

The ultimate purpose of the modelling of groundwater dynamics was the detailed evaluation, in a stochastic framework, of the solutions determined by the DAT as optimal. The stochastic framework provides the expected salinity response of the system as well as the responses within an uncertainty range, e.g. $\pm\sigma$. It is finally worth noting that the SUTRA model embedded in the dynamic programming screening module of DAT used the same conceptualisations, initial conditions and physical parameters as the detailed analysis, using only a coarser grid for computing efficiency. Tests showed that the results of the refined and of the simpler model runs differed only slightly from one another. Prieto (2001) gives a compact account of the groundwater dynamics work.

4.1.2 *The engineered system*

- Desalination.

The know-how of desalination technology (scientific/theoretical state-of-the-art, as well as design and economic aspects) was compiled. The compilation is based on data found in the literature, on field data collected from over 100 industrial plants in Spain and on laboratory data from pilot plant experiments. Further processing of this information gave approximate relationships that hold in the salinity range of brackish water (S = 1,000–10,000 ppm TDS). These relationships concern the flow recovery ratio,

R (S) = (freshwater produced)/(saline water input)

$$R(S) = 0.8291 - 2 \times 10^{-5}\, S \qquad (3)$$

and simple cost functions for the operational cost per m^3 of water produced by reverse osmosis and for the total production cost, both as functions of salinity, S, and of daily flow of water produced, Q_p (m^3/d):

$$C_{O\&M}\,(€/\mathrm{m}^3) = 8 \times 10^{-6}\, S + 0.395 - 0.0194\, ln\, Q_p \qquad (4)$$

$$C_{prod}\,(€/\mathrm{m}^3) = 10^{-5}\, S + 0.7413 - 0.0436\, ln\, Q_p \qquad (5)$$

In addition, a programme was integrated in DAT with which the user can carry out a generic plant design and a detailed determination of the operational and capital costs of that plant. As a rule of thumb, desalting of brackish water of 5,000 ppm TDS costs 50% of desalination of seawater. Of course, production conditions change continuously as the relevant technologies evolve. However, given that energy costs are about one half of total operational costs and twice as high for desalting seawater than for brackish water, the energy aspects will continue to weigh heavily in favour of brackish water desalination.

- Wastewater treatment and re-use.

The cost of treatment of wastewater to various standards (denoted by T_i) was determined and summarised in cost functions that were integrated in DAT. Cost curves for investment and for operation and maintenance (O&M), as functions of plant capacity, were prepared that correspond to levels of wastewater treatment satisfying effluent quality standards for four final destinations (Fig. 8). These destinations are as follows: T_1 = sea outfall (no *sensitive* receptor), T_2 = irrigation (USEPA guidelines), T_3 = surface spreading for aquifer recharge (guidelines of California State Department of Health Services) and T_4 = direct injection for aquifer recharge (USA Drinking Water Standards). Incremental unit cost for upgrading a conventional plant (T_1

or T_2) to the level required for aquifer recharge (T_3 or T_4) is obtained by subtracting the relevant cost curves. An empirical function for the wastewater plant surface as function of plant capacity was also integrated in DAT. Costs associated with land requirements were not taken into account because they range widely. Finally, DAT also includes functions developed for the approximate calculation of pumping costs.

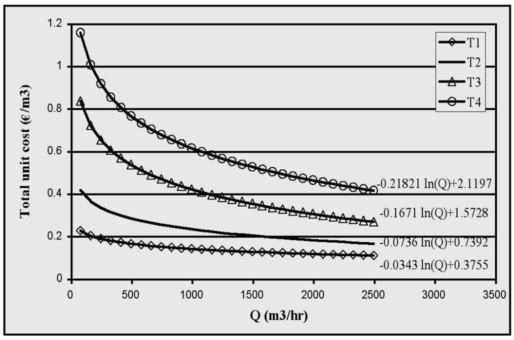

Figure 8. Total unit cost for the four levels of treatment studied.

4.1.3 *Decision-aid procedure*

All aspects of the problem have been integrated in a modular decision-aid tool. DAT is a Visual Basic application that helps to execute the analyses shown in Figure 9. The complexity of the procedure dictated that the optimisation be carried out in a screening stage and in a final design and cost analysis stage. These stages are summarised here; details are given in Koussis (2001). DAT is capable of launching numerical simulation codes such as the FORTRAN screening code. The CPU-intensive Monte Carlo simulations were executed off-line. DAT has also a limited capability to link to the GIS software MAPINFO. The DAT modules concern:

1) *Water demand:* Estimation of potable water demand, during the planning period, and of the monthly wastewater volume generated that can be used for recharge, based on population, number of visitors, economic activities, etc.
2) *Desalination:* Encapsulation of the model for desalination technology, which provides, among others, the required maximum daily capacity of the desalination plant, based on the freshwater demand and on the salinity of the feed water.
3) *Wastewater contribution:* Determination of the available re-cycled water (% of freshwater use) and of its quality and of wastewater treatment costs; estimation of the *agricultural value* of the re-cycled water that is not used for recharging (and is thus available for irrigation).
4) *Screening of scenarios:* Given the natural recharge rate and the pumping rate calculated by the desalination module, the level of salinity for different profiles of recharge with treated waste- water is estimated for 20 years. Profiles that cause groundwater quality deterioration are assessed an environmental penalty (*external cost*), based on the conditions after 20 years. Screening applies dynamic programming to find the optimal flow rates of the recharge well (for a given set of pumping and recharge locations), using minimum cost as the objective function, subject to meeting the water demand and to certain salinity limits. The screening procedure determines the operational plus external system cost of alternatives, for pumping/recharge rates and sites, and ranks them according to cost.
5) *Economic evaluation:* The Net Present Value (NPV) is calculated for user-selected scenarios, which can be all or certain of those identified by screening. The detailed analysis concerns detailed design of the engineering facilities, refined modelling of the aquifer dynamics in a stochastic framework, and economic evaluation of each solution, considering discount rates and water prices. Thus, apart from the investment, O&M and environmental costs, the estimated revenues take into account the total water supply (fresh aquifer water and desalinated water).

DAT can be also used in a simulation mode to carry out NPV sensitivity analyses on variations of certain physical or economic parameters such as the discount rate or the cost of energy. These analyses can be done only for alternatives identified by the screening module. However, the user can examine additional alternatives off-line, by manipulating the output files of the optimisation step; these files contain the pumping and recharge schedules and the associated pumped water salinity for each alternative solution. Again, the salinity time series are obtained by executing SUTRA runs off-line. This

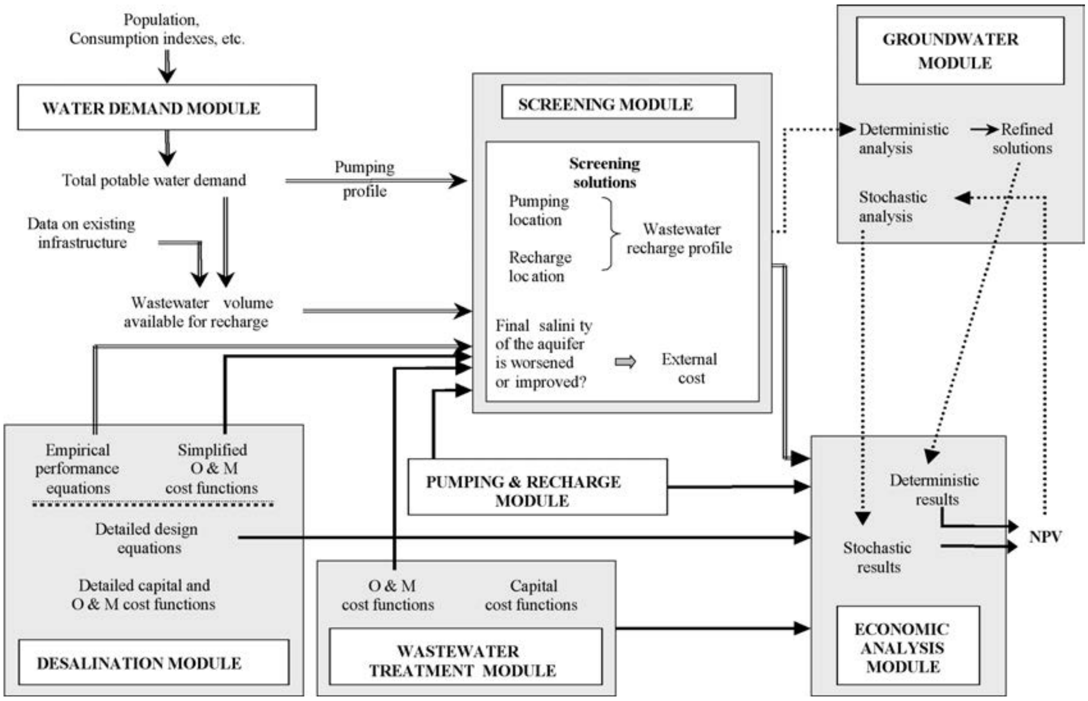

Figure 9. Optimisation procedure used in WASSER: screening and final design and cost analysis stages.

approach can be used, for instance, in order to escape from local minima traps to which the optimisation procedure may lead. The non-convexity of the cost function (due to non-linearity of the flow and transport equations) used in the dynamic programming procedure may cause local optima. The external cost is a penalty assessed in the 20th year by the screening procedure, if final salinity in the aquifer is increased over the initial levels; a gain is assessed for a salinity reduction. This external cost is assumed to be equal to the total cost of electricity required to desalt the affected water volume.

5 APPLICATION OF THE WASSER CONCEPT: RHODES CASE STUDY (Koussis 2001).

Refinement, practical tuning and demonstration of DAT were undertaken in three pilot case studies, chosen to include aquifers of different characteristics and water supply systems exposed to different conditions of demand stress. These aquifers are the Coastal Aquifer of Israel and two aquifers in the islands of Rhodes and Cyprus that are currently exploited rather heavily, experiencing salinity problems. Presently, desalination of seawater or water transfer is considered as means of meeting the rising water demand in these locales. Here, we outline the Rhodes case study as one specific example of the DAT application.

5.1 *Introduction*

This case study concerned the Tsairi basin situated in the northeastern part of the island of Rhodes. There are about 45 wells in the basin, of which 30 are in active use. Three quarters of the total water volume is produced by municipal wells and is used to meet the domestic demand of the city of Rhodes and of the communities of Koskinou, Trianta and Pastida, all located outside the basin, as shown in Figure 10. There are also private wells that produce water to satisfy local demand for agricultural or domestic and hotel-business use. For geochemical reasons, the aquifer freshwater has already a relatively high salt content: TDS concentration in inland areas unaffected by the sea is around 500 ppm, with corresponding chloride (Cl^-) concentration of

70–80 ppm. Based on historical and recent groundwater quality data, seawater intrusion has already rendered the water from wells located within 1 km from the coast unsuitable for drinking, while it has only recently started to affect the quality of wells further inland.

Three municipal wells, located about 3 km from the coast, produce over 40% of all water pumped from the aquifer and ~55% from the zone within 4 km from the coast. These wells are regarded as critical for the water supply of the Tsairi basin; hence only the lower part of the basin was considered in the conceptual model and in the analysis of groundwater dynamics. The model comprised the boxes shown in Figure 10: a 1 × 1 km coastal zone, already affected by sea intrusion, and a 2.5 km-wide × 2 km inland zone that might be affected in the future.

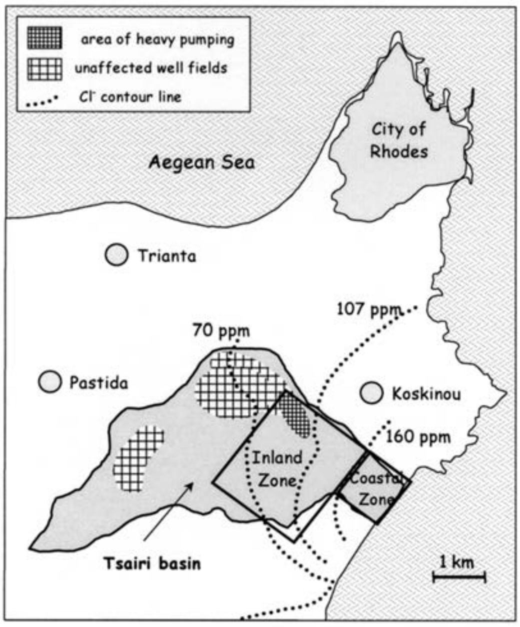

Figure 10. Rhodes case study: Tsairi basin with model boxes, well fields and salinity (Cl^-) contour lines.

5.2 *Water demand*

The demand that these wells must satisfy (required pumping) was estimated from projections of the greater area's total water needs. Figure 11 shows the pattern of the demand during a typical year. Notable are the seasonal variability (pumping in the summer is about seven times higher than in the winter; 3/4 of all pumping is done from May to September) and the dominant domestic use (due to tourism). Figure 12 shows the volumes of the estimated water demand and of the re-cycled water considered to be available for recharge during the 20-year planning period. Since all of the treated wastewater is currently discharged to the sea, actually available effluent exceeds that assumed. A 50% re-cycle reserve was kept on the assumption that, should a scheme of wastewater re-cycling be initiated in one basin, interest could arise for similar applications elsewhere.

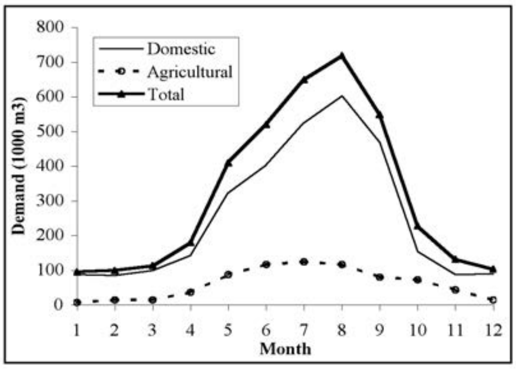

Figure 11. Water demand and available wastewater.

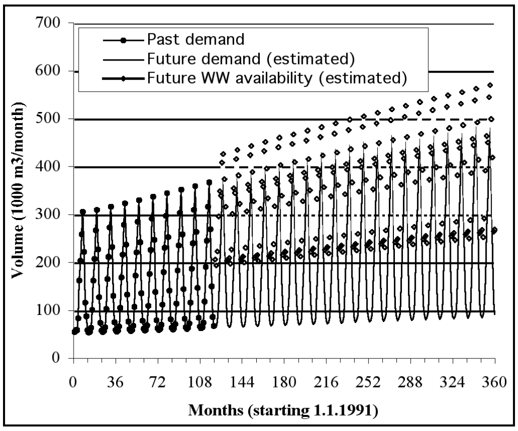

Figure 12. Water demand and wastewater available for recharge.

The objective of the Rhodes case study was to apply the WASSER concept in order to examine optimal ways of satisfying this demand. Seawater desalination was accepted as the only realistic water supply alternative to desalination of brackish water. The salinity of ~5,000 ppm TDS for the groundwater in the inland well was set as criterion of *sustainability*; 1,000 ppm TDS was set as the limit above which the pumped water requires desalination before use (as groundwater of salinity over 500 ppm TDS is already being used without treatment).

5.3 *Surface hydrology and natural aquifer recharge*

The surface hydrology of the Tsairi basin was simulated using the SWAT model (Soil and Water Assessment Tool, Neitsch *et al.* 1999). However, in the basin there are only ephemeral, ungauged watercourses and measured time series of *ET* or groundwater data did not exist. We therefore obtained a rough estimate of the deep infiltration to the aquifer from simulations. The SWAT simulations produced an annual water balance (*closing* to 1%) with an *ET* value (68% of precipitation, *P*) matching that of previous studies (Perisoratis 1992, Iakovidis 1996) and runoff (17% *P*) and infiltration (14% *P*) values that are reasonable for the hydro-morphologic and hydro-geologic conditions of the basin. Measurements of daily precipitation and max and min air temperature at the Rhodes airport for the 10-year period 1988–1997 were used in the simulations. The mean annual precipitation of this data set is 628.2 mm and the corresponding natural recharge (N_R) of the aquifer 86.8 mm/yr. The precipitation and the corresponding aquifer recharge for the 10-year simulation period are shown in Figure 13, which indicates almost zero N_R during dry periods.

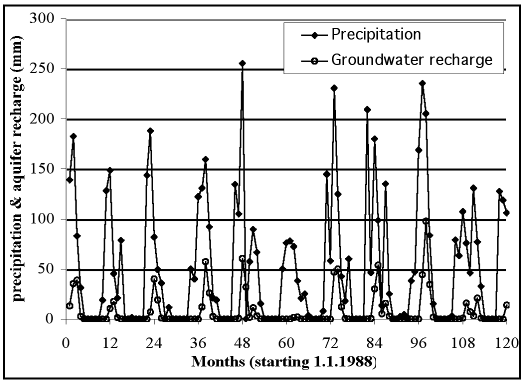

Figure 13. Precipitation and simulated recharge for 1988–1997.

Rainfall time series were generated with the ClimGen weather generator (Nelson 1996). The 10 years of measured daily precipitation and max and min temperatures were used as input for ClimGen, to calculate the mean monthly parameters required for weather generation. 101 precipitation series, of 20-year duration each, were then generated with ClimGen. These became input for subsequent SWAT runs that produced corresponding groundwater recharge time series. The mean statistics of the resulting N_R time series were: annual mean = 86.2 mm, standard deviation = 50.5 mm. In the investigation of the WASSER concept with DAT the *deterministic* solution was obtained from a time series of natural groundwater recharge with statistics close to the set's ensemble statistics (85.6 mm and 53.8 mm). The remaining 100 N_R random realisations were used in the stochastic groundwater dynamics analysis that is discussed below.

5.4 *Groundwater modelling*

An aquifer section, which was schematised and conceptualised as typical for the region, was used in the SUTRA 2-D model. Two conceptual production wells were used in the simulations for obtaining relevant initial conditions: one representing the coastal group of wells and one the inland zone. The distance from the coast and the screen depth of the two wells were computed as corresponding flow-weighted averages of the actual wells of each zone: The coastal zone well was then placed 200 m and the inland zone well 2.7 km from the coast. The mid-point of the well screen was taken as pumping depth. It was further assumed that the future water demand that would be satisfied by the conceptual wells should follow the current distribution of total pumping from the Tsairi basin, 6% from the coastal well and 49% from the inland well. The initial condition was estimated to reproduce the average observed salinity at the conceptual inland well. The initial transition zone was obtained as follows:

1) A transient simulation was made, with initial groundwater table slope 0.015 seawards and initial salinity 500 ppm TDS everywhere in the aquifer, except at the nodes on the sea boundary, where it was 35,000 ppm TDS. Natural recharge was uniform at 0.086 m/yr; pumping and artificial recharge were zero. The simulation was halted when the salinity at the coastal well was ~800 ppm (observed salinity in 1991),

yielding initial pressure and salinity conditions for the second transient simulation.

2) This simulation included pumping at 1991-rates from the coastal and the inland wells, in addition to the mean natural recharge (86 mm/yr). The simulation was stopped when the salinity at the coastal well reached 2,500 ppm TDS (observed salinity in 1999), yielding initial conditions for the third transient simulation. Since salinity of 2,000–3,000 ppm TDS was unacceptable, however the coastal well was closed at this time and the corresponding pumping added to the inland well for the third simulation.

3) This simulation included natural recharge and pumping from only the inland well (at the rate of the original coastal plus the inland well rates) and ran until the well salinity was ~600 ppm TDS (observed average salinity at the group of inland wells in 1999). The resulting salinity and pressure values were the initial conditions used in simulations with projected pumping and artificial recharge rates for the period 2001 to 2020.

The schematic groundwater model for the Rhodes study site shown in Figure 14 illustrates the simulation domain, the assigned boundary conditions, the locations of the point sources and sinks (indices R and P) and the homogeneous *K*-values used in the deterministic simulations. For each investigated scenario, the screening model provided the location of the recharge well and the projected monthly pumping and artificial recharge schedules (Q_P and Q_R) used in the predictive deterministic and stochastic simulations. Table 1 lists the site-specific data and assigned parameter values and the general physical parameter values used in the SUTRA simulations.

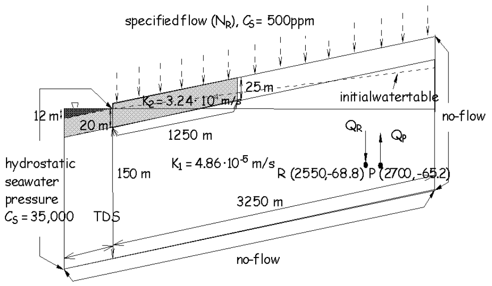

Figure 14. Schematic cross-section of the Tsairi basin aquifer.

Table 1. Physical and numerical parameter values used in the SUTRA simulations for the Rhodes case study.

Parameter	Value
Width of the cross-section (m)	2,500
Mean specific yield	0.36
Freshwater density (kg/m^3)	998.57
Base solute concentration (ppm TDS)	500
Number of elements	3,710
Number of nodes	3,886
Spatial discretisation (m): x-horizontal	$\Delta x = 25$
y-vertical: for elements within 25 m below the upper boundary and 20 m below sea bottom,	$\Delta y_1 = 2.5$
y-vertical: for the rest of the elements	$\Delta y_2 = 10$
Longitudinal dispersivity (m)	12.5
Transverse dispersivity (m)	1.25
Molecular diffusivity of solute in fluid (m^2/s)	10^{-9}
Fluid compressibility	0
Fluid viscosity (kg/m/s)	10^{-3}
Aquifer matrix compressibility	0
Parameter *a* in van Genuchten equation (m·s^2/kg)	5×10^{-5}
Parameter *n* in van Genuchten equation	2
Residual saturation	0.3
Seawater density (kg/m^3)	1,024.45
Density change with concentration coefficient (kg^2/kgTDS/m^3)	750

The conceptual model, initial conditions and all input parameters, except the hydraulic conductivity *K* and the natural aquifer recharge N_R, were the same in the stochastic and in the deterministic simulations. In the generation of 100 equally probable realisations of the spatially random *K*-field for the stochastic simulations, the geometric mean, K^G, was set equal to the constant *K*-value used in the deterministic simulations. Similarly, the total freshwater volumes provided by the mean N_R throughout the simulation period in the spatial-plus-temporal stochastic simulations were equal, or very close to the total freshwater volumes of natural recharge in the deterministic simulations. The deterministic simulations represented therefore the mean hydraulic conductivity and natural recharge of the stochastic simulations.

In this case study, the specific statistics of the log-normal hydraulic conductivity field $Y = ln[K^G(\text{m/s})]$ and of the exponential autocorrelation function were: *mean* $Y = -9.93$, *var* $Y = 0.5$, i.e. moderate degree of heterogeneity everywhere, and anisotropic heterogeneity (integral scales: horizontal 100 m, vertical 40 m; four nodes per integral scale). For similar spatial heterogeneity conditions, comparative analysis of stochastic and deterministic groundwater simulations yielded similar main results for the Rhodes as for the Israel case study. However, the results of the Cyprus case study were somewhat different, primarily due to different aquifer depth and temporal randomness characteristics, as discussed in more detail below.

The deterministic evolution of salinity in the pumped groundwater did not, in any of the three WASSER case studies, exactly reproduce the expected (E) salinity obtained by the stochastic simulations. Figure 15a shows the specific differences between deterministic (dashed line) and expected salinity in the Rhodes case study, with the expected salinity calculated for the two different stochastic assumptions: that of only spatial (K) randomness (solid line) and that of coupled spatial-temporal (K and N_R) randomness (dash-dot-dot line). In the Rhodes case, the addition of temporal randomness (random N_R) to random spatial heterogeneity (only K random) led to only a small increase in the 20-year averaged standard difference between deterministic and expected concentrations, relative to that with only spatial randomness considered. Also, the corresponding salinity standard deviation (SD) shown in Figure 15b, as well as the coefficient of variation ($CV = SD/E$, relative uncertainty) shown in Figure 15c, followed patterns similar to the expected concentration results. This means that the spatial-plus-temporal randomness did not increase much the uncertainty associated with the concentration in the pumped groundwater, compared to the case of only spatial randomness.

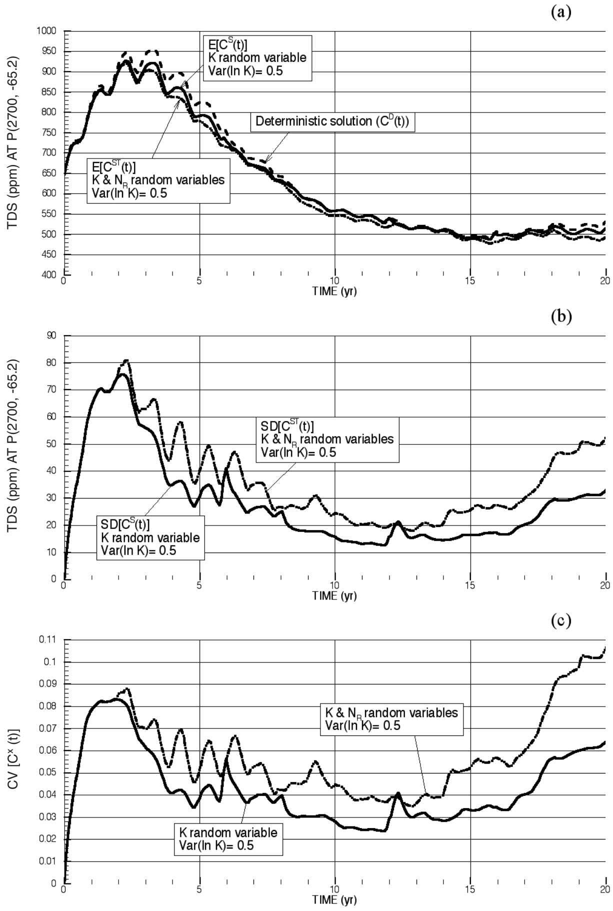

Figure 15. Comparison of salinity statistics in pumped groundwater: a) expected salinity from the spatial (solid line) and the spatial-plus-temporal (dash-dot-dot line) stochastic simulations, and the salinity from the deterministic simulation (dashed line); b) salinity standard deviation from the spatial (solid line) and the spatial-temporal (dash-dot-dot line) stochastic simulations; c) coefficients of variation from the spatial (solid line) and the spatial-plus-temporal (dash-dot-dot line) stochastic simulations.

Comparison with the other two WASSER case studies for the same conditions of spatial randomness shows consistent results for the Israel case. The Cyprus case study, however, exhibits considerably larger increase of the 20-year-averaged standard difference between expected and deterministic salinity. Moreover, it exhibits considerable increase of the salinity prediction uncertainty, as quantified by SD and CV values for the spatial-plus-temporal randomness compared to spatial randomness only. One reason for these differing results is that in the Rhodes (aquifer depth ~150 m) and the Israel (aquifer depth ~100 m) case studies aquifers were deeper than in the Cyprus case (mean aquifer depth ~50 m). In turn, greater aquifer depth implies more capacity to dampen the fluctuation effects of N_R variability. Additionally, in the Cyprus case study, temporal salinity fluctuations were further increased by temporal randomness in the inflow through the upstream boundary added to that in N_R, whereas in the other two cases only N_R was considered temporally random.

5.5 *Investigation of alternative aquifer management schemes*

The investigations concerned recharge of the aquifer with treated wastewater, to combat sea intrusion, and desalination of groundwater pumped from the inland zone, should it become brackish, provided that this was cheaper than alternative water supply options. Since the bulk of the current pumping in the zone of interest is done in three municipal wells located near the centre of gravity of pumping, 2.7 km from the coast, it would not be practical to consider a different pumping location, since this would imply moving these critical wells. Therefore, no alternative pumping locations were investigated, leaving only the recharge-well location to be optimised. Only the injection option was considered for recharge, since the aquifer is rather deep (pumping from the municipal wells is done at 60–80 m depths; the conceptual well was located at 60 m.b.s.l.), while land availability is limited and land prices are high.

Based on the approximate economic data that are applicable to the technical components of the system (see Section 3.1.2), i.e. desalination, wastewater treatment and recharge, and to the external cost, the screening module of DAT identified one optimal solution, hereafter called Solution No. 1. In that analysis, the quality of the available treated effluent is T_1 (secondary treatment) and thus the estimated wastewater treatment cost was for upgrading to T_4 (drinking water).

Solution No. 1 is to continue the current practice of pumping to satisfy the demand, without performing any wastewater recharge, but to desalinate the water once it becomes too brackish to use (for the deterministic natural recharge case, this occurs after 5 years). The graph in Figure 16a shows that Solution No. 1 is acceptable, but marginally so, because the pumped water salinity at the end of the planning period exceeded slightly the limit of 5,000 ppm TDS. The detailed economic analysis by DAT showed that this solution requires a water price of ~0.73 €/m^3 to be viable economically and, in any case, is preferable to seawater desalination.

We also investigated aquifer management under the assumption of zero wastewater treatment cost. This implied that the quality of the effluent of the municipal wastewater treatment plant had to be upgraded, irrespective of its use for aquifer recharge. In this case, the solutions identified by screening included recharge and the optimum recharge schedule depended on the location of the recharge well. By recharging near the pumping well, sea intrusion could be controlled to maintain the salinity of the pumped water low. Solution No. 9, depicted in Figure 16b, is the best in the family of *recharge-solutions* and required no desalination: wastewater was recharged down-gradient of the pumping well at 2,550 m and pumped water salinity was kept below 1,000 ppm TDS at all times.

The detailed economic analysis by DAT determined that the increase of the external cost could also shift the economics towards the recharge-option, even bearing the full wastewater treatment cost, i.e. the cost of upgrading the effluent quality from T_1 to T_4 before recharge. For instance, Solution No. 9 becomes

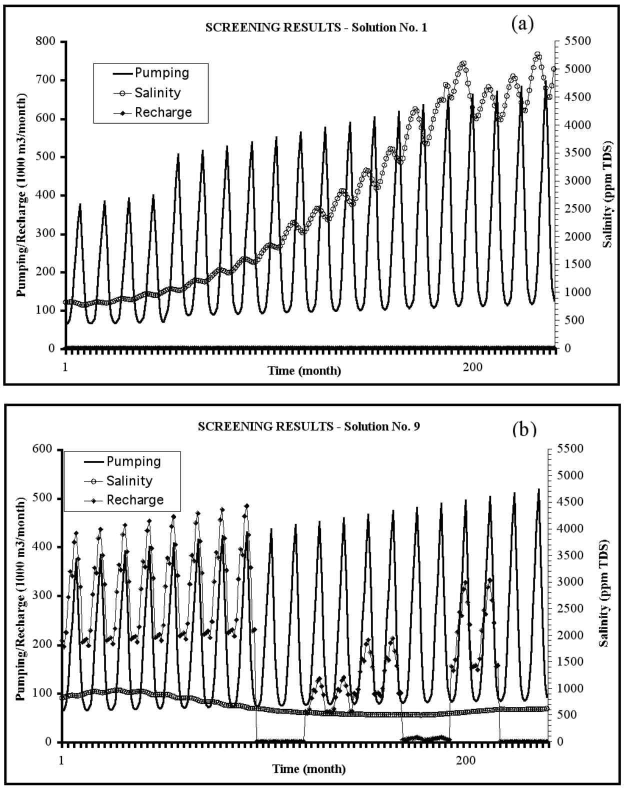

Figure 16. Pumping, recharge and salinity profiles of solutions generated and rated as optimal by the screening model.

economically superior to Solution No. 1 once the external cost is tripled (i.e. calculated with 0.39 €/kWh instead of 0.13 €/kWh).

Taking into account the uncertainty associated with the evolution of salinity (see Figure 16b and stochastic analysis of groundwater dynamics in Section 5.4), Solution No. 9 required a water price of at least ~1–1.30 €/m^3 (depending on the externalities) to be economically viable. Solutions involving wastewater recharge downgradient, at 1,850–2,350 m, also required desalination after 2004–2005. This increased their production cost and they were thus ranked at the lowest positions and always worse than seawater desalination, unless high externalities were combined with low discount rates. In addition, these solutions required a water price of at least ~1.30 €/m^3 to be economically viable. Seawater desalination required a water price over ~1.20 €/m^3 to be economically viable, but even for this water price it was inferior to several solutions requiring wastewater recharge and/or brackish water desalination.

The screening module's Graphical User Interface allows visualisation of the impact of a management scheme on the progression of sea intrusion, providing a monthly step-by-step picture of the salinity of each grid cell. This aids in understanding the reasons behind the *choices* that the screening code makes. Figure 17 compares the progression of sea intrusion in the middle and at the end of the planning period for the two solutions of Figure 16. In Figures 17 and 18, the light bullet indicates the pumping well and the dark bullet the recharge well; the sea is on the darker, left part of the graph.

Figure 18 shows the extent of sea intrusion at the end of the 20-year planning horizon for some management solutions that were identified as optimal by the screening programme, for the same physical and economic parameters but different recharge well locations. Examination of Figures 16 and 18 elucidates the implications of the selection of the planning horizon and of the importance given either to *present water use* (i.e. to the salinity of the pumped water) or to

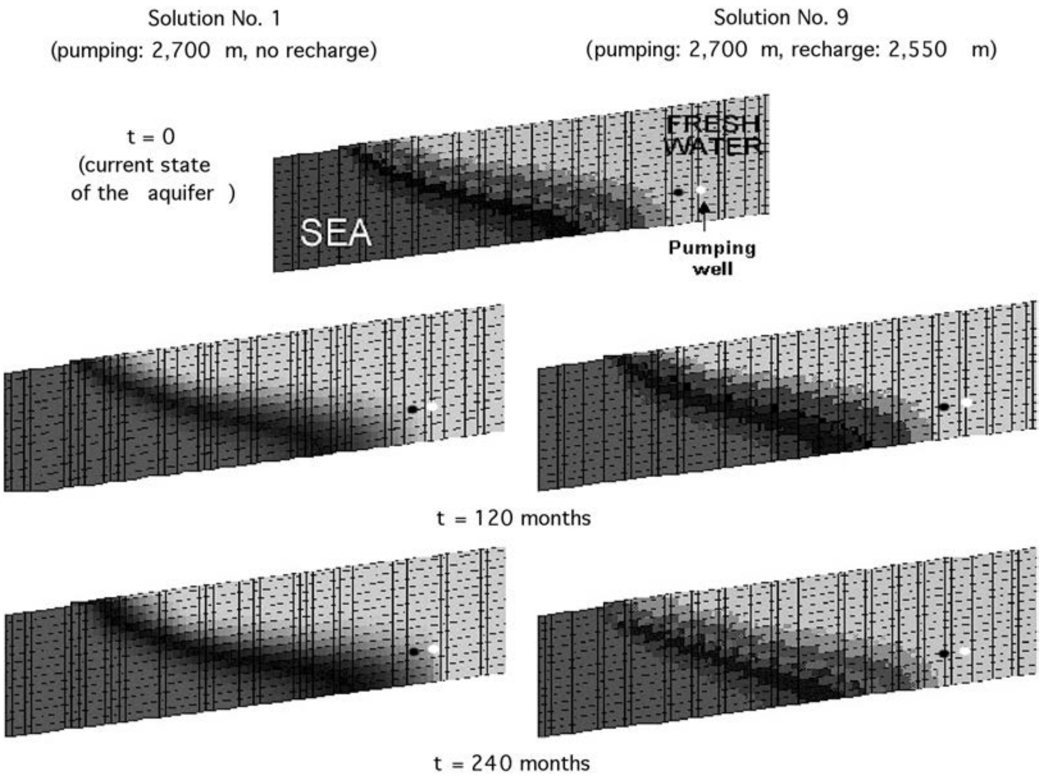

Figure 17. Rhodes case study: Seawater intrusion at t = 120 months = 10 years and at t = 240 months = 20 years, for the alternative strategies of solution No. 1 and solution No. 9.

overall state of the aquifer. When the quality of the pumped water weighs more than the salinisation of the aquifer, schemes that imply, even temporarily, higher salinity at the well are penalised relative to schemes that imply a worse aquifer state after 20 years and might thus be less sustainable.

For instance, alternatives involving recharge closer to the coast (top graph in Fig. 18) could not emerge as *best*, either during the screening stage or during the detailed economic analysis, even with externalities tripled. And this happens, even though they may seem to be more effective in controlling sea intrusion and could be conceivably more economically viable for a longer planning period. The reason is that such solutions result in a higher short-term salinity of the pumped water, since recharge over the current seawater/freshwater interface pushes some volume of seawater toward the inland well.

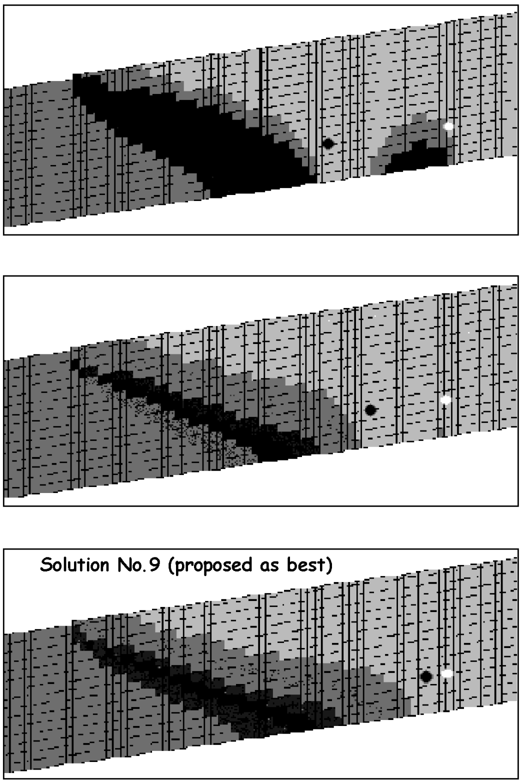

Figure 18. Graphs of the salinity field in the aquifer, at t = 20 years, resulting from alternative management schemes that were identified by the screening module as optimal for different locations of the recharge well.

5.6 *Discussion of the solutions, and assessment and outlook for WASSER*

The case study of Rhodes showed that the premise of WASSER can lead, indeed, to sustainable schemes of groundwater development that are economically more favourable than seawater desalination. The Cyprus and Israel case studies, which were omitted due to space limitations, confirmed this finding. Among the Rhodes solutions, two alternative strategies were found to be the most interesting:

A. *Low externalities*: Solution No. 1 in Figure 17. Let the aquifer respond to the evolution of pumping, without artificial recharge, and desalinate brackish water when the salinity at the pumping well exceeds the limit of 1,000 ppm (occurred at year 2005). Note that the salinity increased also at other locations of the aquifer.

B. *High externalities:* Solution No. 9 in Figure 17. Perform intensive wastewater recharge from the beginning, thus keeping the salinity of the pumped water below 1,000 ppm during the entire period.

The level of assumed externality influences significantly the selection between strategies *A* and *B*. In this study, the calculation of externalities was arbitrary in the selection of the cost of electricity and also incomplete, as only a part of direct externalities were quantified. For example, loss of income, due to the discontinuation of an economic activity resulting from the degradation of groundwater quality, was not quantified. Nevertheless, since strategy *B* has a better economic performance than strategy *A* in the case of higher externalities, it was considered a more sustainable water supply policy. In any case, the degree of internalisation will remain a largely political matter. Furthermore, the results obtained should be re-considered in view of the evolution of cost figures for desalination and for wastewater treatment. It also stands to reason that the choice of a specific salinity sustainability limit influences the optimisation procedure and its outcome. For example, a strict salinity limit will favour use of recharge to keep the salinity of the well water low and a solution viable.

We emphasise also that the DAT optimisation runs concern past climatic conditions and probable realisations of climate scenarios. As these may not occur in the future, the simulated recharge and pumping schedules do not constitute prescriptions for management strategies for the next 20 years. Strategies for near-actual time management can be developed on the basis of monitoring, of short-term climate predictions and of model simulations.

On a more general note, it can be said that the economics of brackish water desalination and of advanced treatment of secondary wastewater effluent to drinking water standards are not very dissimilar, in principle. Therefore, the *first best* solution proposed by an optimisation that evaluates the WASSER concept on the basis of minimum cost would tend to favour *single-technology* solutions, i.e. either only recharge or only desalination, *whenever this is possible*. [The Rhodes exercise confirmed this outcome, however the optimal solution for the Cyprus case study combined recharge and brackish water desalination]. Thus in retrospect, one can argue that further investigation of the WASSER concept should seek to incorporate examination of system variability and related uncertainty already in the screening stage, not in the stage of detailed economic analysis (with sensitivity analysis option), as done in the present version of DAT. This is sensible because uncertainty is the very reason that might render a *single-technology* solution infeasible.

Finally, management of aquifer resources on the basis of operating a dual-plant (i.e. one capable of desalting brackish and seawater) for the desalination of brackish water, with the sea as reserve source, and of aquifer recharge with treated wastewater can increase system reliability at little added cost. This option should be included in a future version of DAT.

6 CONCLUDING REMARKS AND OUTLOOK

Coastal zones are often characterised by high population densities and intense economic activities. Such conditions inevitably place heavy demands on the finite water resources of a coastal ecosystem in general and on its aquifers in particular. Evidence of strain on coastal aquifers is the, at times extensive, seawater intrusion that has occurred under the intensive development of groundwater in many areas around the world. Coastal aquifers are also vulnerable to contamination from the landside, e.g. due to nitrates derived from fertilisers and septic tanks and to pesticides; this vulnerability is often heightened by the proximity of aquifers to the surface. Yet such threats are not unique to coastal locales.

For this reason, we emphasised in this chapter more large-scale aquifer contamination from intruding seawater in response to intensive groundwater development. This threat is more acute in semi-arid regions, where overdrafts are more likely to be used as a means of dealing with periodic water shortages. It is worth recalling in this context that mixing 2% of saltwater with fresh groundwater suffices to render aquifer water unsuitable for drinking.

In an era of increasing competition for access to fixed resources, it is realistic to expect a continuing pressure on the resources of the coastal aquifers. Then, given the risks of pollution and the high economic, social, health and environmental costs, the issue is to make this development sustainable. On the technical side, while there is no substitute for sound hydrology and hydraulics and for detailed monitoring, more attention should be paid to assessing prediction uncertainty and implied risks. Uncertainty may concern the forcing of the system (e.g. due to climate change or to demand/demographics), the hydrologic system itself and its characterisation (e.g. due to aquifer heterogeneity or to monitoring errors and insufficiencies), and the evolution of technology costs. On the awareness side, the state authorities should educate the public, through information campaigns starting already at the school level, to appreciate water scarcity better. On the institutional side, conservation and proper use of water should be promoted, also via economic instruments such as water pricing (*the user pays*) and penalties for degradation of water quality (*the polluter pays*).

With precipitation-derived resources fixed, alternate water sources must be increasingly considered. Two non-traditional sources, with cost-effective potential, are brackish groundwater, which can be treated to potable quality standards, and treated wastewater, which can be recycled as groundwater to meet various demands. The economics of desalting of brackish water are favourable and re-use of wastewater can be applied more widely when the public's perception improves. The application of a management scheme that is based on these options has been demonstrated here with a case study of a coastal aquifer in Rhodes.

APPENDIX: ELEMENTS OF DYNAMICS OF SEAWATER INTRUSION INTO AQUIFERS

A few results of elementary hydraulics of saltwater intrusion into aquifers in response to well pumping are added, keeping mathematical formulations simple to allow highlighting certain important concepts analytically. Thus, our 2-D (profile) example makes use of the Ghyben-Herzberg sharp freshwater-saltwater interface approximation, which is based on the Dupuit theory for nearly horizontal flow. Following Ghyben-Herzberg, the sharp interface, $\zeta(x)$, with x a horizontal co-ordinate, is related to the freshwater potential referenced to the sea level, $h_f(x)$, by the hydrostatic balance across the interface:

$$\zeta(x) = \frac{\rho_f}{(\rho_s - \rho_f)} h_f(x) = \frac{h_f(x)}{\delta s} \quad ; \quad \frac{1}{\delta s} = \frac{\rho_f}{\rho_s - \rho_f} \tag{A1}$$

The total thickness of the freshwater lens floating on the stagnant seawater is then

$$H_f(x) = \zeta(x) + h_f(x) = h_f(x)\left(1 + \frac{1}{\delta s}\right) = h_f(x)\left(\frac{1 + \delta s}{\delta s}\right) \tag{A2}$$

Following van Dam (1999), we consider an infinitely long island of width L underlain by a deep unconfined aquifer of hydraulic conductivity K that is recharged at the rate N_R, as shown in Figure A1.

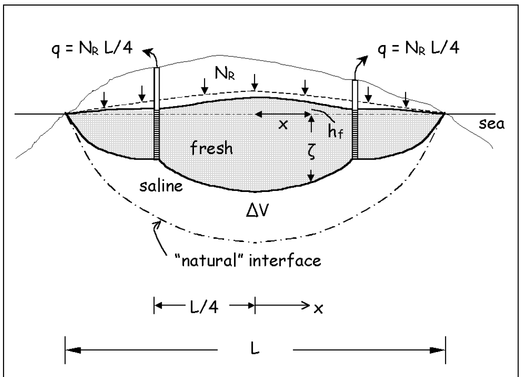

Figure A1. Freshwater lens in *infinitely long* island: pre- and during-development by 2 lines of wells. Adapted from van Dam (1999).

The free surface of the freshwater lens in $0 \le x \le L/2$ ($x = 0$ at the island centre line) is the parabola

$$h_f^{\,2}(x) = \left[\frac{N_R L^2 \delta s}{4K(1 + \delta s)}\right]\left(1 - x^2/L^2\right) \tag{A3}$$

Equations A2 and A3 show that the maximum thickness of the freshwater lens depends more on the width of the island L and less on the recharge rate N_R. Now, if we want to abstract a fraction α of the total recharge $N_R\,L$, i.e. $q = \alpha\, N_R\, L$, by means of a line of wells at $x = L/2$, the free surface parabola is

$$h_f^{\,2}(x) = \left[\frac{fL^2\delta s}{K(1+\delta s)}\right]\left[\left(\frac{1}{4} - \frac{x^2}{L^2}\right) - \left(\frac{q}{N_R L}\right)\left(\frac{1}{2} - \frac{x}{L}\right)\right] \tag{A4}$$

From Equation A4 follows that if, e.g. half of the recharge is abstracted, $q = N_R\, L/2$, the shape of the freshwater lens is such that the interface reaches the surface at the well lines. Thus saltwater enters in the well screen. On the other hand, abstracting $q = N_R\, L/2$ by means of two well lines at the quarter-points of the island width yields a much more favourable shape of the freshwater lens ($L/4 \le x \le L/2$), namely

$$h_f^{\,2}(x) = \left[\frac{fL^2\delta s}{K(1+\delta s)}\right]\left[\left(\frac{1}{4} - \frac{x^2}{L^2}\right) - \left(\frac{q}{N_R L}\right)\left(1 - \frac{2x}{L}\right)\right] \tag{A5}$$

Placing the wells closer to the coastline reduces the *upconing* of seawater by capturing more recharge, which keeps the abstracted freshwater volume small. This simple analysis underlines the importance of proper configuration of the abstraction system. $T = \Delta V/\Delta F$ is a measure of the characteristic transition time from one state of dynamic equilibrium to another due to a change in system forcing ΔF ($L\, \Delta N_R$ or q) that causes a change in volume ΔV.

A saltwater cone is created underneath a pumping well. However, by Equation A1 and for $\rho_s = 1{,}025$ kg/m^3, $\rho_f = 1{,}000$ kg/m^3, the saltwater cone mirrors the cone of depression of the freshwater potential near the well, but is magnified 40 times. Thus for a unit lowering of the freshwater potential the saltwater rises 40 units (yet the steep interface reduces the accuracy of the Dupuit assumption and hence of the Ghyben-Herzberg approximation).

Dagan & Bear (1968) have solved the problem of saltwater upconing underneath a point or line sink located in the vertical plane at a distance d above the initial, undisturbed interface. They used perturbation analysis and assumed infinite fresh- and saltwater domains to derive formulas for the interface evolution. For a point sink of strength Q, the interface equilibrium position exactly underneath the sink is

$$\zeta_{sink} = Q/2\pi\delta\ sKd \tag{A6}$$

From Equation A6 follows that the pumping rate limit Q_{max} that maintains an interface rise $\zeta_{sink} \leq d$ is

$$Q_{\max} \leq 2\pi K d^2/\delta s \tag{A7}$$

However, theoretical (Muskat 1946) and most experimental data reviewed by Motz (1992) show that the interface becomes unstable at $\zeta_{sink} \sim d/2$ and starts to rise rapidly, reaching the well bottom. Motz derived an analytical solution for interface upconing that also assumes a small rise of the sharp interface, but accounts for a well screen of finite radius partially penetrating an anisotropic aquifer of finite thickness. The maximum pumping rate criterion of Motz is more complex than Equation A7 and showcases the importance of anisotropy. Wirojanagud & Charbeneau (1985) derived an analogous design criterion based on numerical solutions and dimensional analysis. The operation of coastal collector wells that avoids invasion by saline water is a matter of practical importance. For this reason the critical pumping rate is often limited to less than $Q_{max}/2$, yielding $\zeta_{sink} < d/2$, e.g. $\zeta_{sink} \sim d/3$. This drastic reduction should be interpreted also from the realistic perspective of a dispersed salinity field. Fresh- and seawater mix gradually, forming a salinity field with rapid variation under pumping wells; the sharp interface can be thus viewed as, e.g. the contour line with 50% of the salinity of seawater. The paper of Panday *et al.* (1993), its discussions by Motz (1995), Dagan (1995), Charbeneau (1995), and Panday's reply (1995) add insight to the problem, also bringing to light the intricacies of numerical modelling. Detailed (at the local-scale) modelling analysis of the freshwater-saltwater interaction is a prerequisite for a realistic assessment of management scenarios. Sorek & Pinder (1999) survey the currently available relevant codes.

ACKNOWLEDGEMENTS

On behalf of the WASSER consortium, the authors wish to express their gratitude to the European Commission's Directorate General for Research (DG XII) for the support of this research, under contract ENV4-CT97-0459, and to the General Secretariat for Research and Technology of Greece for providing co-funding.

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