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Controlling seawater intrusion by treated wastewater recharge. Numerical modelling and cost-benefit analysis (CBA) at Korba case study (Cap Bon, Tunisia)

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ABSTRACT

Treated wastewater (TWW) reuse for managed aquifer recharge (MAR) is becoming an important measure for integrated water management in areas with water scarcity. Among reuse applications, interest in aquifer recharge to control seawater intrusion in coastal aquifers is growing worldwide. At the Korba-Mida aquifer (Tunisia), local aquifer recharge with treated urban wastewater has taken place through three infiltration ponds since 2008. An *ex post* Cost–Benefit Analysis (CBA) is presented to assess the recharge impact at the groundwater level in a coastal aquifer after 3 years of recharge. A MODFLOW-based groundwater numerical model was developed to guide the impact assessment. The local model results showed that the recharged volume was slightly higher than extractions. The economic results indicated that the internal rate of returns accounted for 14.46%, while the discount rate of project investment was 4%. According to the sensitivity analysis, this project is feasible for the present wastewater treatment cost (0.1 TND per m^3, Tunis Dinar) and up to 0.25 TND per m^3. Possible effects on groundwater quality as an added influential final externality cost were not considered.

Keywords: Seawater intrusion; Artificial recharge; Treated wastewater reuse; Cost-benefit analysis

1. Introduction

The Mediterranean region faces an increasing water scarcity trend because population, tourism (especially in coastal areas) and agricultural activities are on the increase, precipitation diminishes, and there is no fully optimised water resource policy in force. When meeting water demand, many coastal aquifers have been over-exploited by pumping in excess of recharge. This implies significant water deficits, a drop in the water level of some aquifers that go below sea level, and higher chloride concentration in wells from seawater intrusion (www.semide.net/topics/groundwater/Mediterranean_Groundwater_Report_final_150207.pdf).

In response to water scarcity, Water Authorities have taken measures to increase quality water supplies and to regulate demand. Policies include water transfer, the conjunctive use of surface and groundwater, groundwater banking, recycling and reusing wastewater, desalination, improving water use efficiency, implementing economic instruments such as water pricing and water trade, including water purchases for environmental purposes, among others [1]. Reuse

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applications are gaining popularity because the quality of treated waste water (TWW) discharges constantly improves, which is in line with increasing technology and the need for alternative or non-conventional water sources to meet water demands. Among the techniques to increase water supply or to control seawater intrusion in coastal areas, managed aquifer recharge (MAR) schemes [2,3] with numerical simulation guidance [4] are an interesting alternative. Supplementing natural recharge with MAR can increase water levels and improve water quality as this practice can limit seawater intrusion into aquifers. The Intergovernmental Panel on Climate Change (IPCC) [5] presents desalination as a potential option, together with wastewater reuse, to adapt to climate change impacts, especially in arid and semi-arid regions. A wide range of methods are used for MAR to meet local conditions. For further information, readers are referred to www.iah.org/recharge/.

For most water resources projects, economic analyses that consider all economic costs and benefits are needed [6,7]. To demonstrate their economic feasibility, certain indicators are applied, such as a cost-benefit ratio. A cost-benefit analysis (CBA) is a relatively simple and widely used technique that assesses how a particular market or economy at a specific site may be changed by new policies and practices [8,9]. This method is also useful to become more fully aware that the wider economic effects that a specific project implementation may have are important. For groundwater issues, this description usually entails predictions about the consequences of man-made actions and economic implications for groundwater systems. CBA was used herein to assess the relative desirability (in economic terms) of competing management alternatives to address pollution problems. There are two general types of CBA analyses: *ex ante* and *ex post*. An *ex post* CBA of a single purpose project for groundwater recharge, therefore after completing a project, is illustrated here.

To solve water shortage in specific areas, the cost of producing desalinated water has been studied in detail [10,11] because, in theory, desalination costs are easy to assess from the data provided by operators; published studies that have focused on MAR are quite scarce [12,13] and available economic information is still limited. This research has two objectives: to estimate and discuss the environmental impact of an artificial recharge system (MAR) in a salinised coastal aquifer through a groundwater flow model; to economically assess recharge. The main objective was to examine the economic and environmental costs and benefits (CBA) of recharge (R) with treated wastewater (TWW) by infiltration ponds based on the obtained field results and on information provided by operators. The economic analysis required forecasts of costs and benefits, and conclusions could differ depending on such forecasts. A sensitivity analysis was calculated to obtain a range of results based on possible values.

2. Study area

The Korba area is located in the Cap Bon peninsula in north-east Tunisia (Fig. 1), covers over 500 km^2 and is limited by the Mediterranean Sea on the eastern side. The region has a semi-arid climate characterised by average annual rainfall of 480 mm and seasonal high variability. The most important aspect of the precipitation pattern is rainfall concentration in a few days. Mean annual temperature is 20°C, and ranges from 14°C (December, January and February) to 27°C in August. For the area where recharge takes place and the study period [14], the average values at the Korba meteorological station were 520 mm of rainfall, 1300 mm of ETP and a temperature of 20.5°C.

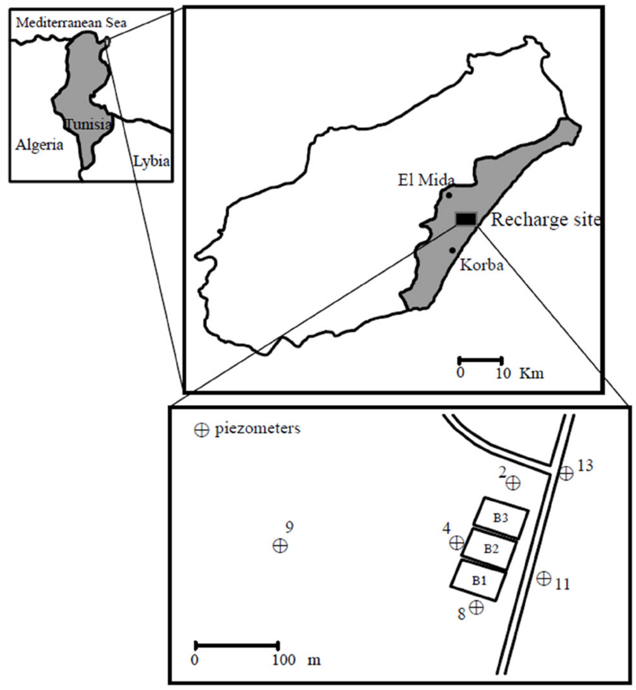

Fig. 1. The Cap Bon Mida study area and the location of the artificial recharge facility.

The population in this region is about 120,000 inhabitants. Tourism has grown considerably, particularly during seasonal periods when the population can double. The main economic activities are tourism and irrigated agriculture (horticulture, viticulture, fruit-growing, grain farming and livestock), with principal crops of strawberries (300 ha), potatoes (1200 ha), tomatoes (3500 ha), peppers (3500 ha) and other vegetables (1500 ha) (data provided by INGREF). Although agricultural activities, along with some agro-industries dominate, host food industries, textile, dairy and paper industries are also present. Water demand for irrigation is met mainly from the Plio-Quaternary aquifer exploitation and by surface water from the Medjerda-Cap Bon canal. In the last decade, the region has seen a new water demand trend to support a growing tourist population.

The Cap Bon peninsula is a basin filled with marine sedimentary materials of Tertiary and Quaternary ages [15]. From a geological standpoint, three main geological formations of Miocene, Pliocene and Pleistocene ages are identified (Fig. 2a).

The Middle Miocene is the lower part of the Beglia formation of detrital deposits. The upper part is composed of sandstones and marls with lignite levels and clays. The Upper Miocene (the ”Somâa sands” unit) only outcrops in the southern part of the study area. Pliocene sediments,

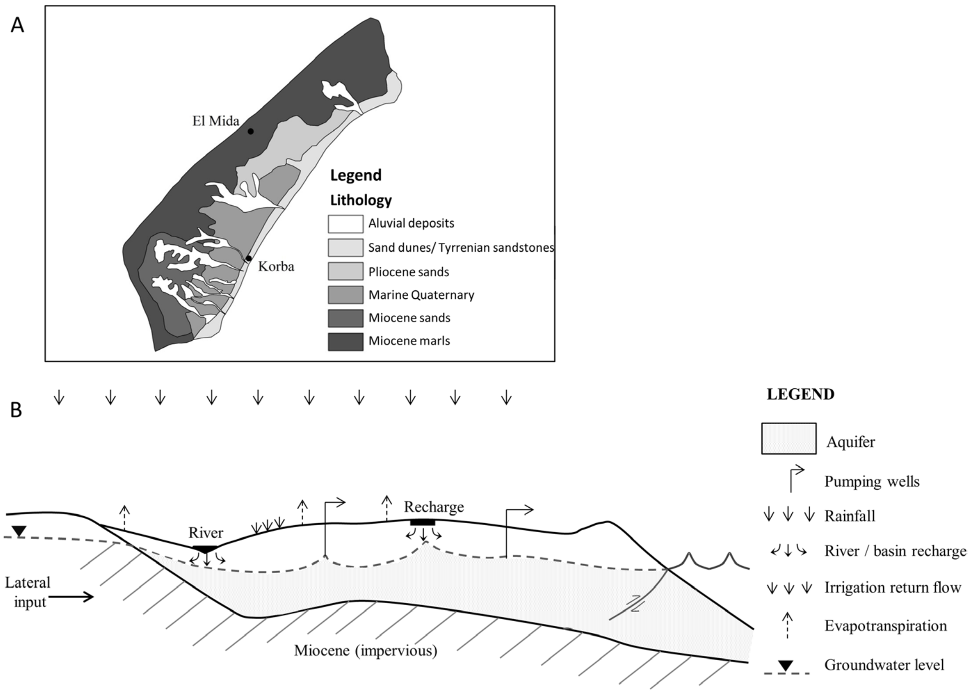

Fig. 2. A) A geological sketch of the study area and B) A schematic diagram of the Korba-Mida conceptual model illustrating hydrologic flows and boundary conditions.

unconformably deposited over the Miocene, are composed mainly of alternate layers of sandstone-sand-marl with clayed sandstones on top. The Pleistocene consists of continental deposits with alternating sands, sandy clayed and clay levels. The upper continental Pleistocene is composed of reddish silt that overlies a carbonate unit and a calcarenite whose thickness ranges from m to cm. The Tyrrhenian carbonate sand-bar covers a length of about 1.2 km along the coastal border. Late Holocene deposits are constituted by recent alluvial sediments of Chiba Wadi, sebkha deposits and sand dunes.

The Korba-Mida aquifer is composed of Quaternary, Pliocene, and Upper Miocene deposits (Fig. 2b). It constitutes a strategic resource to guarantee supply and has been well-studied and monitored. The groundwater abstraction of the 1960s, done mainly for irrigation purposes, started at the Korba-Mida aquifer through over 9,500 wells, and has lately reached 54 hm^3/year. After four decades of aquifer exploitation, the groundwater level has dropped, which has led to seawater intrusion in the coastal area of Korba. According to hydrogeological reports, in 2004 withdrawn groundwater accounted for 135% of recharge [16], which has led to depression cones at the groundwater level in the central part of the aquifer area.

In an effort to mitigate the impact of high-chloride water on groundwater supplies, local agencies, led by the local Water Authority, have implemented strategies that have involved the conjunctive use of TWW and groundwater to meet demand. Since 2008, MAR from TWW was introduced as part of Integrated Water Resources Management (IWRM) to preserve groundwater resources, meet water demand, cope with water scarcity and control seawater intrusion at the Korba-Mida aquifer.

2.1. Groundwater hydrology. The Korba-Mida aquifer

The Marine sedimentary Korba-Mida aquifer unconformably lies on the middle Miocene bedrock composed of impervious marls or marine origin. It constitutes the aquifer basement (Fig. 2a and 2b) and extends over a coastal string of 420 km^2, skirted by the Mediterranean Sea and Miocene impervious marls to the west. The hydrostratigraphic system, composed of Plio-Quaternary and Miocene formations, has been described in detail by a number of researchers [17–22].

The confined Miocene aquifer is hydraulically connected to the Plio-Quaternary by fractures and geologic discontinuities. However, the relationships between both Miocene and Plio-Quaternary aquifers are not clearly understood. The Pliocene, semi-confined locally due to less permeable deposits [14,23], is composed of an alternate of stratigraphic layers that contain freshwater and brackish-water. The average thickness is 85 m, which can be as much as 150 m in the south area, and 30 m in the north and the study area. The aquifer is recharged primarily by precipitation where the aquifer outcrops, and by infiltrated water from existing streams, with additional contributions from irrigation return flows. The aquifer discharges on the east margin to the sea. Nowadays, the coastal sabkhas (wetlands) no longer constitute the natural outlets of the Tyrrhenian hydroestratigraphic unit due to the reversal of the hydraulic gradient by groundwater abstraction. Aquifer exploitation by wells is approximately 51.8 Mm^3/y; data obtained by analysing groundwater abstraction data in 2026 wells and for the 2008–2011 period (database inventory provided by INGREF).

The regional groundwater flow direction is NW–SE from the topographically high areas to the Mediterranean

Sea. At the recharge site, the water table ranges from less than 1 m below the surface, to below sea level in the eastern part, while groundwater level oscillations between dry and wet seasons may reach up to 3 m. According to previous studies [22] and *in situ* tests, hydraulic conductivity (k) values can range from 10^{-6} to 10^{-3} m/s, and aquifer average porosity is 0.12. From the hydrochemical point of view, the groundwater in the aquifer is generally alkaline, hard to very hard, and fresh to brackish, with salinity concentrations between 36,000 and 800 mg/l [24].

2.2. Artificial recharge facility

The recharge site is located about 100 km southwest of Tunis City and approximately 300 m north of the Korba wastewater treatment plant (Fig. 1) [25,26]. In this area the aquifer is composed of homogenous alluvial deposits of fine to medium sand with some gravel deposits. The hydrogeological parameters are relatively uniform throughout the area; permeability is estimated between 1.6×10^{-3} and 2×10^{-3} m/s. Soils are constituted by 80% sand, 15% loam and 5% clay. Texture ranges from sand to sandy loam.

The Korba wastewater treatment plant, which can treat 7500 m^3/d, began to operate in July 2002; presently, it receives around 5000 m^3/d. Plant treatment consists in a pre-treatment, a secondary treatment by an oxidation channel process, and a tertiary treatment of maturation ponds. The site design includes three infiltration ponds of 50 m × 30 m × 1.5 m (B_1, B_2, and B_3) that cover a surface area of 4,500 m^2 and the recharge setup consists of two infiltration basins that function simultaneously with a daily injected volume that varies from 533 to 886 m^3 (3–8 h) in each basin. The supply system comprises a basin buffer tank with 300 m^3 capacity, and is gravity-fed by a 400-mm diameter pipe that is automatically controlled by a valve.

Altogether, the system operated during the study period (2008–2013), except for December 2012, when the system did not operate for technical reasons. At the end, the TWW applied in the three infiltration ponds accounted for 1.7 Mm^3, and the water table in the monitoring piezometers increased by around 1 m (on average).

3. Methodology

3.1. Data collection

For study site characterisation purposes, hydrologic information was collected from available reports, publications and the existing databases of the administrations in charge of water (General Direction of Water Resources, Ministry of Agriculture in Tunisia (DGRE) and Regional Commissariat for Agricultural Development (CRDA) of Nabeul). The meteorological data in data sets included rainfall, potential evapotranspiration; cultivated crops and irrigation dose, soil distribution maps, geological logs, hydrogeological parameters and piezometric levels, and were provided mainly by the National Institute of Research in the Rural Engineering of Water and Forestry (INRGREF-Tunis). The obtained spatial data were compiled in a database based on a GIS (Arc view 2.3). No new data sets were generated to characterise the subsurface hydrology of the study area, which was based on existing information.

The groundwater level was monitored by the Water Authority (Direction Générale des Ressources en Eaux) through 36 wells and 16 piezometers drilled to a depth of 7–20 m at the study site. The daily monitoring of the TWW that infiltrated in the aquifer was done in three piezometers (2, 4 and 8, Fig. 1), located next to the infiltration ponds through an online data capture system. Monthly monitoring by INRGREF-Tunis included groundwater level, and also the groundwater physico-chemical parameters (electric conductivity, temperature and pH). *In situ* monitoring started in December 2008 (before the recharge) and continued until December 2013 (867 days), which accounted for 60 field surveys over 5 years.

3.2. Modelling approach

Local groundwater MAR modelling, which focused on groundwater level changes, required the development of a regional numerical flow model for the Korba area, which involved extending the modelling domain by approximately 420 km^2 (Fig. 3). This model provided the boundary conditions, flow direction, sources and sinks required for modelling the local recharge area. Having defined and simulated the initial groundwater conditions on a regional scale, a local-scale model was developed to simulate artificial recharge. A steady-state model was used because the temporal coverage of the hydrological data sets needed in this stage was lacking, and also to obtain some indication of the aquifer system response to artificial recharge.

Hydrogeological data, including permeability and storage coefficient values, were obtained from field research, existing databases, and a number of studies and publications previously conducted in the Korba area [16,27–29]. To reduce the number of free parameters, and based on lithology and references, porosity was assumed to be 0.12 for the entire modelled area. K values (m/s) were spatially distributed in the regional model along three zones with values of 6×10^{-5} (North), 3×10^{-4} (Central) and 4×10^{-5} (South).

On both the local and regional scales, the conceptual aquifer system model consisted of two independent layers. Layer 1 represented the upper aquifer (average thickness of 40 m); a phreatic aquifer consisting of Plio-Quaternary deposits with an upper limit on the topographic surface and the base defined by Miocene impermeable marls (Layer 2). Hydrological stresses included natural and artificial recharge (rain, rivers and ponds), lateral inflow/outflow and groundwater pumping.

For the numerical model, Visual MODFLOW.PRO 2009.1. was applied [30], based on the VISUAL MODFLOW 3.0 code. A two-dimensional flow was simulated, and the model was run under steady-state conditions to estimate the piezometric level data for the regional and local models in 2001 [24]. Hydrological stresses included natural recharge from rainfall and irrigation return flow, lateral inflow and outflow, groundwater pumping and artificial recharge. For the regional model, a grid of 400 × 400 m (126 rows and 85 columns), aligned with the aquifer boundary piezometric level over the area of interest (Fig. 3) with the upper limit, corresponded to the land surface based on a 90 × 90 m DEM topographic map (www.cgiar-csi.org). The bottom layer was set at the impervious Miocene layer depth. The depth of the lower boundary condition was obtained from hydro-

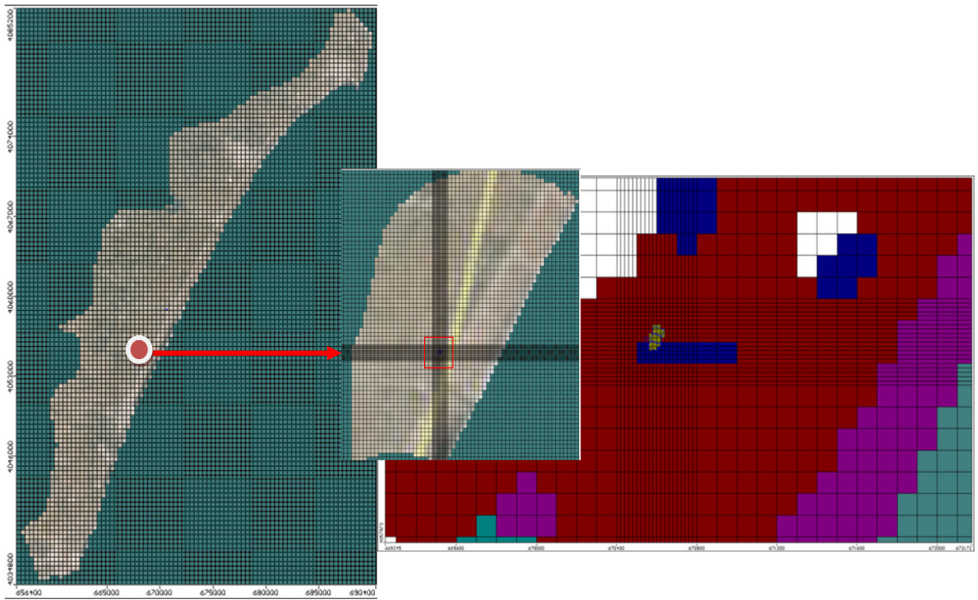

Fig. 3. Regional and local model definitions and grid overlay.

stratigraphic information obtained from wells and piezometer geological logs.

The boundary conditions of the aquifer system were based on the 2001 piezometric level observations and the natural geological or hydrogeological boundaries (Fig. 4). Constant head conditions for Layer 1 were defined based on the topographic surface and the Mediterranean Sea coast ($h = 0$). Based on the observed groundwater flow and the impervious nature of the geological materials boundary, no flow boundary conditions were assumed for the remaining aquifer boundaries. The river boundary conditions were dependent on the state of the aquifer. For the lower boundary, a no-flux boundary condition was assigned.

The groundwater recharge from precipitation and irrigation in the outcropping aquifer, the uppermost active cells, was calculated for the 2008–2012 period by Visual BALAN 2.0. This is a user-friendly modular design model based on a physical water-soil balance that has been successfully applied in many areas [31,32]. For this study, six recharge zones were assumed to be homogeneous compared to the soil parameters, land use (forest, urban, irrigated and non-irrigated crop areas), slope, altitude and meteorological data (Fig. 5). In the recharge ponds area, a value of 80 × 10^3 mm/y (data provided by INGREF) was allocated to only one cell of 400 × 400 m. Outputs were estimated from the information available from about 2026 pumping wells.

The model was calibrated (or a model contrast was made due to the limitations imposed by lack of data) through trial-and-error by matching the modelled and measured piezometric levels over 35 wells. Model calibration was quantitatively assessed by calculating the determination coefficient (R^2), where a value of 1 represents a perfect fit between the observed and predicted values, and the root mean square error (RMSE), whose optimal value was zero, of the simulated and measured hydraulic heads [33].

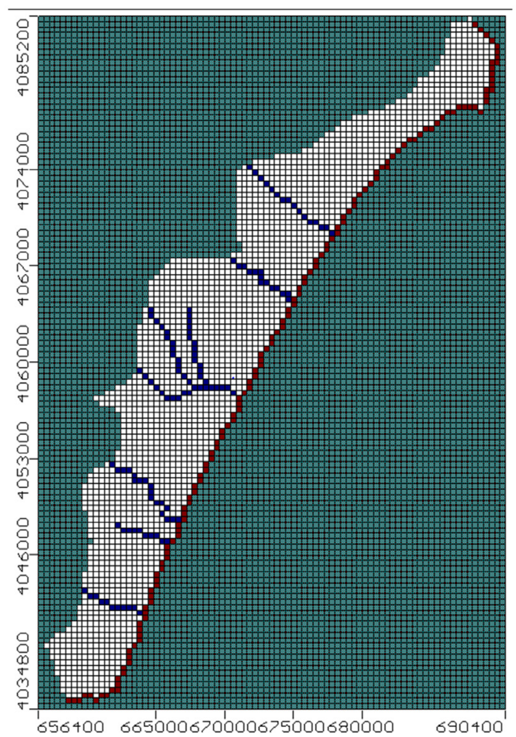

Fig. 4. The Korba-Mida regional model extension, including the boundary conditions. Grey: no flow; blue: river boundary conditions; red: constant head conditions.

The calibration/contrast process involved assessing the hydraulic conditions, hydraulic parameters and running the steady-state model.

The second step in the modelling process involved the aquifer surface area, where artificial recharge took place

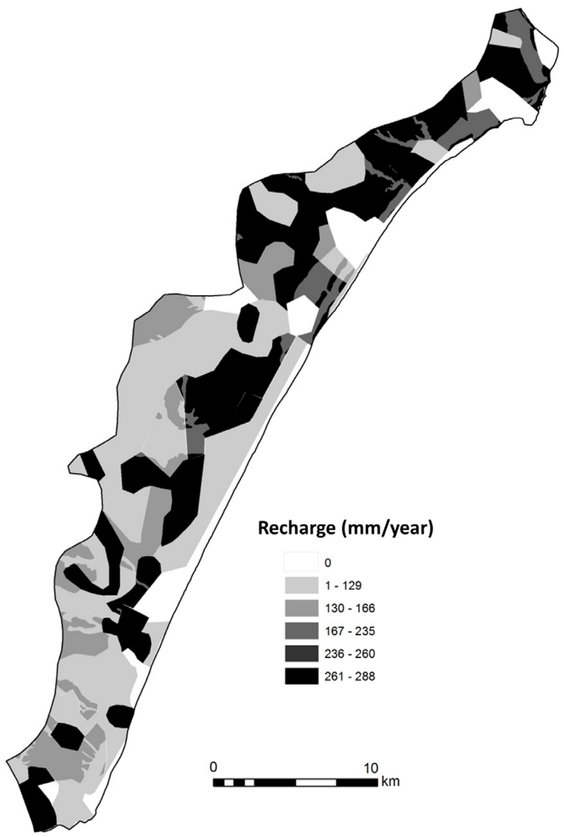

Fig. 5. Recharge zones definition (precipitation plus irrigation return) obtained with Visual BALAN 2.0.

over approximately 57 m^2. For the local-scale model, a variable grid (between 100 × 100 m and 20 × 20 m), made up of 86 columns and 86 rows, was superimposed on the horizontal plane, where the upper aquifer surface corresponded to the land surface elevation (Fig. 4). The recharge basins input (1500 m^3/day; data provided by INGREF) was simulated in 17 cells of the 20 × 20 m grid. The boundary conditions derived from the regional model outputs (the boundary condition according to the obtained piezometric level). One main objective of the modelling exercise was to also estimate the aquifer response to changes in aquifer recharge (reclaimed surface) via a second scenario, which included two sets of recharge ponds at the facility site. The hypothetical recharge field was located north at an approximate distance of 1000 m from the existing field, and the model was rerun by replacing the recharge input amount.

3.3. Cost-benefit analysis (CBA)

In order to assess the economic analysis, costs related to water supply, infrastructures, management and operation, environmental costs and beneficial effects were estimated based on the information provided by the operator.

For the CBA analysis, the alternative "*groundwater damage to be avoided by recharge*" was compared with the "*do-nothing*" option, which implied lack of groundwater salinity control under the 'without artificial recharge scenario' situation. In this study the analysis of economic feasibility was based primarily on estimates of direct land benefits and the costs derived by improving salinity concentration in soil and groundwater [34].

The Net Present Value (NPV) criterion is the principal investment project evaluation criterion of summarising [6] the values of economically relevant costs and benefits during the project life span. The mathematical expression of the NPV of benefits for the two scenarios, '*do-nothing*' and '*recharge-R'*, was computed by the following equation (1).

$$NPV = \sum_{0}^{n} \frac{D_t - D_t(R) - C_t}{(1+r)^t} \quad (1)$$

where NPV: Net present value of net benefits. For NPV ≥ 0, the project is considered with benefits; D_t: Damage if no action is undertaken (without R) in year t; $D_t(R)$: Damage if action is undertaken (with R) in year t; C_t: Cost of R in year t; t: The relevant year; n: Years involved in the economic horizon (life span); r: Discount rate (or interest rate paid for using borrowed funds)

The comparison of costs and benefits, to occur at different time points, must be expressed as a common measure, and the mechanism is known as discounting. The rate at which benefits or costs are discounted is known as the discount rate. The applied discount rate was 4% per year according to the cost of borrowing money for financing development projects; the amortisation period was 30 years. This period selection was based on the project's economic life according to previous experiences from around the world and the pay-out period for funds to build the project. As benefits and costs vary from year to year during the analysis period, a conversion to present worth quantities was made by taking into account the discount rate. All the CBA estimations were done according to the local currency, namely the Tunis dinar - TND (1 TND~0.5€).

The recharge impact on the aquifer was estimated by taking into account the area with positive changes at the groundwater level (up-level), and the salinity reduction in the aquifer observed in monitored wells compared to reference values prior to recharge. According to existing hydrogeological information [28], the areal distribution of groundwater salinity (%) at the Korba study site is classified as: a) accounting for 90% of the aquifer extension between 0 and 5 g/l; b) 6% between 5 and 8 g/l and c) 4% greater than 8 g/l, which constitutes agricultural abandoned land. As the effect of recharge had a circular mound shape, which also extended over the sea, only the inland aquifer impact was considered for the assessment (half the circle produced by recharge). For the baseline analysis, a 5-km influence radius of recharge was considered, which corresponded to 3925 ha inland (half of the 5-km radius surface). To support the baseline selection, the groundwater modelling results were considered.

For the '*do-nothing*' option, groundwater salinity was expected to increase from the present situation at a 10% rate in time, while, groundwater salinity in the '*recharge, R*' scenario was expected to rise at an 8% rate for the first 3 years of recharge, 2% to 2021 and only 1% during the project life span; i.e., 30 years. Salinity reduction implied an increase in the land economic value in the local market price.

The Internal Rate of Return (IRR). was also calculated. The IRR is an investment efficiency indicator, and a measure of the comparison made of the two alternatives (*'project with R'* and *'no project at all'*) or the profit rate which the owner receives. To calculate the IRR, the NPV has to be set as being equal to zero and needs to be solved for discount rate *r*.

As the treated water cost and the reclaimed area are critical issues when assessing economic costs, a sensitivity analysis of the CBA results for ten tertiary wastewater treatment costs (from 0.25 to 0.7 TND/m^3) and five reclaimed land surface extensions (corresponding to half the inland surface, estimated from the 5-km, 4-km, 3-km 2-km and 1-km circle radii) was carried out to assess the project's feasibility. Treatment costs included the amount of tertiary treatment needed to remove any nitrate present in TWW.

The modelling results of the recharge impact and groundwater chemical quality data from the sampling campaigns done in the wells were also used for the considered external factors definition.

3.3.1. Costs estimation

The identified 'direct costs' included those needed to implement the project. They were related to investments, annual costs and externalities due to other effects on the economy that the project could cause (Table 1). Costs of the mitigation measures to minimise environmental impacts (negative externality) were also accounted, and they included:

- Investment: project set up and constructed facility costs (land acquisition, engineering, building construction and equipment, [29]
- Infiltrated water: calculated from the total volume of supplied water since the project started and the cost of wastewater treatment (0.1 TND per m^3)
- Operation and maintenance: applies to labour, administration, supplies and replacement of short-lived equipment. Manpower includes the labour resources spent in installing the plant. As the system is gravity-driven, other energy costs (ex. water pumping) do not apply
- Environmental: the effect of the negative impact of groundwater quality by nitrate contamination due to recharge [27] is a negative externality, and the associated cost of removing nitrate contamination from the aquifer needs to be considered. The treatment cost for nitrate removal has been estimated at 0.3 TND per m^3 according to [35]

TWW intake did not imply reducing treated waste water availability for another use. Therefore, no derived effects were considered.

Table 1
Breakdown of costs estimation for the recharge facility

Costs	TND × 10^3
Investment	
Land acquisition	89,200
Building and equipment	671,000
Cost of water	1,353,750
Operation and maintenance (salaries, supplies)	120,293
Environmental (negative externality)	2,707,500

3.3.2. Benefits estimation

As benefit is not the same as revenue, benefits for this project were defined in terms of 'direct user benefits'. Beneficial effects were measured in terms of the damage that the project prevented in soil and groundwater during the 30 years of the project life span compared with '*no project*' damages or increased salinity in soil and water (NTD). An estimation of the present worth benefit was calculated as 8,489.128 × 10^3 NTD.

4. Results and discussion

4.1. Modelling results

Fig. 6a presents the simulated steady-state piezometric heads for 2001 for the regional scale model. The piezometric surface shows similar domes and depression cones to the measured values presented in 2001 [24] while the existing groundwater flow is adequately reproduced, as indicated by the following key features: regional flow is to the sea, where aquifer discharges and depression cones are observed in pumping wells.

As one goal was to provide the boundary conditions for a refined local model, the regional model results were satisfactory, as reflected by the measured values and captured groundwater trend, and no further discussion is needed. Fig. 6b shows a scatter plot of the computed and measured hydraulic head. R^2 was 0.92 and RMSE was 3.07 m. Most wells were between the two 95% confidence intervals, which confirmed goodness of fit. The calibrated hydraulic conductivity was 1×10^{-5} (northern sector), 2×10^{-4} (central) and 5×10^{-5} (southern sector).

The artificial recharge effect on the aquifer water balance on the regional scale is presented in Fig. 7. This figure shows changes in the inflow and outflow components and estimations of the recharged volume of water (Input-R: wastewater, precipitation and irrigation return) and pumped wells (W), as obtained from the MODFLOW outputs. It is evident from Fig. 7 that the recharged volume (52.5 hm^3) was slightly higher than that pumped by wells (51.8 hm^3), while the seawater input in the aquifer decreased. As expected, no significant differences in the lateral contributions between both scenarios were found, and the impact of seawater input-output on water balance was negligible.

For the local scale model, the water level contours, output pre-recharge and after-recharge are shown in Fig. 8a and 8b. The simulated groundwater levels agreed with the field observations. The closed contours at the recharge field location presented a cone with the positive groundwater values following the effect of radial recharge, the 0 m groundwater level was displaced inland and the water table values were between 0 and –0.3 m.a.s.l., and never rose ground surface at the recharge site. The mound height reached 0.9 m, a value that was also assessed in the monitoring piezometers, while the surface extension was around a 1-km radius compared to the '*no recharge*' scenario. As TWW had lower salinity than native groundwater (saline), the recharge of water to the aquifer also resulted in native ground water dilution and in a change in the aquifer's salinity conditions by displacing the intruded seawater of previous years. Groundwater quality improvement through the effect of water

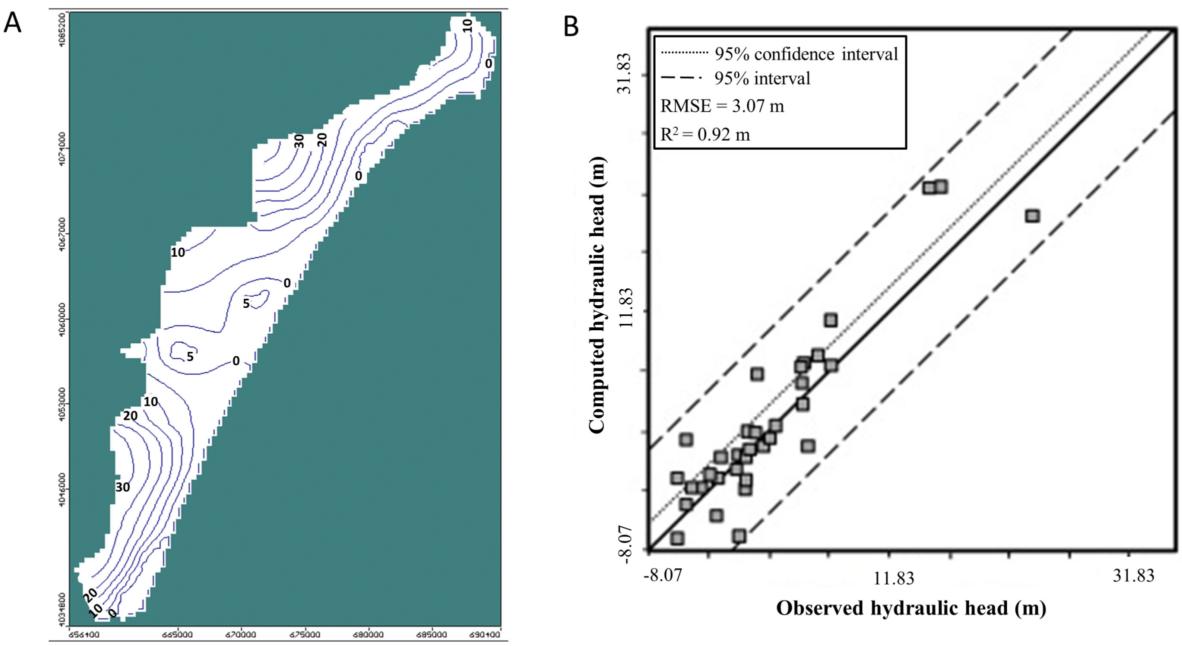

Fig. 6. A) Regional model: simulated piezometric level for 2001 and B) Scatter plot of the observed and estimated hydraulic head by the regional model in 35 observation wells, plus statistics.

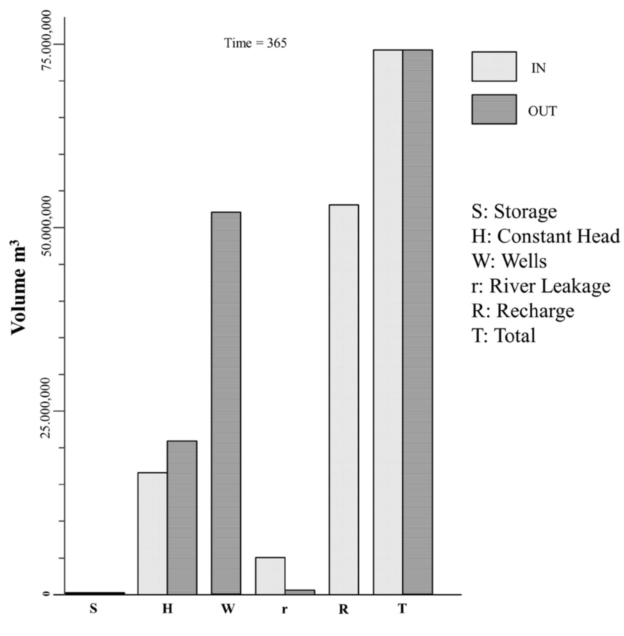

Fig. 7. Aquifer balance results. Regional scale.

mixing movement (TWW and the aquifer mixing front) was observed up to 2 km from the recharge site (accounting for an estimated total surface of around 1,200 ha), according to the observations made at the wells [21,26]. For the second simulation scenario (two recharge fields), the recharge mound effect accounted for 80% of the local-scale simulated area (Fig. 8c).

The artificial recharge effect on aquifer balance is presented in Fig. 9, which shows changes in the inflow and outflow components, and recharged water volume estimations (input: precipitation, irrigation return; TWW is included as R in Fig. 9b) and pumped wells (W). It is evident from Fig. 9b that the recharged volume (3.19 hm^3) was slightly higher than in that pumped by wells (2.74 hm^3), while the seawater input in the aquifer reduced (0 hm^3) and led to decreased salinity, as expected.

4.2. CBA results

4.2.1. Comparison of costs and benefits on present worth

All the direct costs calculated for the Korba recharge facility are shown in Table 2. For the 30-year life span at a 4% discount rate, the present worth cost amounted to 3,166932 × 10^3 TND. The estimated present worth benefit, based on aquifer salinity damage prevention, was 8,489.128 × 10^3 NTD.

The NPV (Eq. 1) for the baseline project shown in Table 2 accounted for 5,141.671 × 10^3 TND, which represents an IRR of 14.46% (much higher than the applied discount rate of 4%).

Regarding the sensitivity analysis, the results of the selected range of possible wastewater treatment costs (0.25–0.7 TND/m^3) versus the reclaimed land surface extension are plotted in Fig. 10. The NVP value varied with the different combinations of reclaimed land and treated water costs. The NPV for the baseline estimation (3,925 ha of inland reclaimed surface) was always over 1 for all the water costs, and also for the 2,512 ha extension. A wide range of project costs was feasible. However in the remaining defined scenarios (smaller reclaimed extension), except for the 1,413 ha (radius: 3 km), the 0.25 and lower TND/m^3 treatment cost, the 2 km radius (1261 ha) and costs lower than the 0.125 TND/m^3 treatment cost, the NPV generally presented neg-

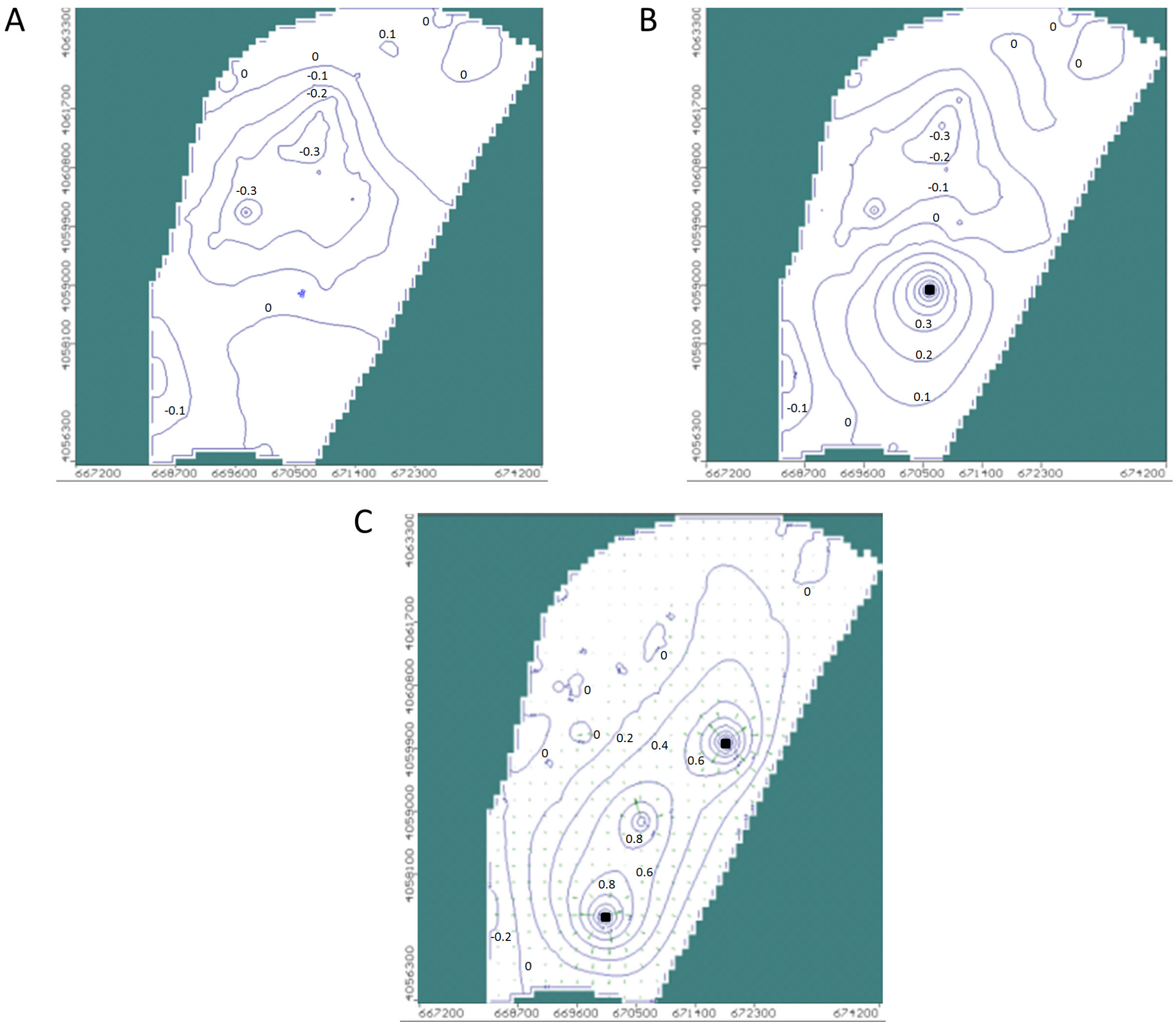

Fig. 8. Local-scale piezometric level: A) without recharge; B) with recharge and C) effect of two recharge fields.

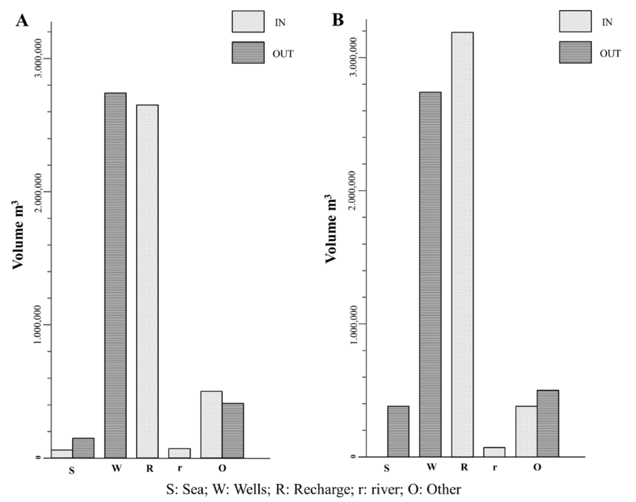

Fig. 9. Local scale and groundwater balance: A) without recharge (pumping: 2.74; recharge: 2.65) and B) groundwater with recharge (pumping: 2.74); recharge 3.19, seawater input: 0).

ative values, and the recharge project did not involve benefits (low feasibility).

The second recharge scenario (two infiltration fields, Fig. 8c), obtained through numerical simulation, indicated that the reclaimed land area would render the project quite feasible.

5. Discussion

The application of numerical models for the economic assessment of MAR projects' implementation and feasibility appears an interesting option for different recharge scenarios, which is the case herein. As shown in previous studies, the application of steady-state models has been particularly useful in situations that required long-term predictions of groundwater changes in environments where stresses and boundary conditions can be reliably considered to be constant over time [36]. Numerical modelling was particularly important for assessing the present recharge impact on the land and aquifer surface at the Korba facility. The obtained results were similar to the outputs presented by other

Table 2
Net present value (NPV) of the project (estimation based on 3,925 ha extension of recharge surface at a 4% discount rate)

Cost present worth (10^3 TND)		Present worth benefit(10^3 TND)	NPV(10^3 TND)
Investment	760,200		
Infiltrated water	781,891		
Operation and maintenance	61,058		
Environmental	1,563783		
Total Present worth cost	3,166932	8,489.128	5,141.671

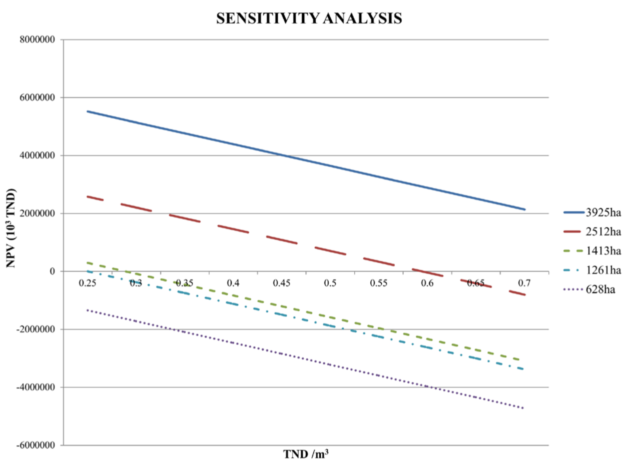

Fig. 10. Sensitivity analysis of the NPV after considering different water costs and reclaimed land surfaces.

researchers, who applied different numerical codes [37,38]. The steady-state modelling approach applied herein was appropriate for simulating hydrological conditions.

The recharge mound dimension (height and lateral spreading) is a response of the water table to recharge, and is governed by basin size and shape, recharge rate, duration and aquifer characteristics. The distance travelled by the salinity front presented a slight trend with recharge distance, caused by continuous mixing waters at different ratios, which was greater than that observed and for the simulated groundwater level mound, as supported by the field measurements of salinity in wells [21,25]. The implication of flushing TWW into the aquifer is that the salinity in wells becomes a mixture of two water sources, but a better quality than the present groundwater, which radially moves outwardly from the recharge basin due to dispersion and mixing. Consequently, the simulated results are an indicator of the hydrodynamic process governed by the hydraulic principles that take place in the porous media by recharge. The dispersion and mixing from different flows through porous media is a complex hydrochemical process. To improve the understanding of the process, and to comprehensively assess the possible impact on quality changes, require salinity mixing-based specific mass transport numerical simulation.

For the baseline analysis (reclaimed surface that accounted for 3925 ha), the IRR of 14%, was higher than the discount rate (4%) and the NPV was always over 1, which indicates its high feasibility. The IRR equates the present worth of benefits and costs, and is a measure of the comparison made between the two alternatives (*'project with R'* and *'no project at all')* or the profit rate which the owner could receive. The data provided by a study to trace sewage in salinised groundwater by [21], along with the evolution of the groundwater quality results of [26], indicate that the displacement of the mixing front of groundwater and recharged waters would only affect an area of a 2-km surface radius of influence approximately (1420 ha). This is much lower than the baseline project estimations and limits the project feasibility for a wastewater treatment cost of up to 0.125 TND/m^3, which is slightly higher than the actual cost (0.1 TND/m^3). It is noteworthy that other possible effects on groundwater quality, which could lead to undesired or unknown long-term impacts [39,40], were not taken into account as an added externality.

The results from the two infiltration field-simulated scenarios revealed that under uniform recharge conditions, the mound increased to a greater horizontal extent by occupying around 80% of the reclaimed area. For this scenario, the surface extension attained by recharge would render the project highly feasible. Nevertheless, the increase in the associated treatment cost for nitrate pollution removal is expected to be higher than the baseline scenario due to the amount of volume provided by recharge. The increased cost is expected to be compensated by the benefits made by the amount of new reclaimed land and aquifer. It is important to note that the new present worth costs for the two recharge fields, and associated benefits, have to be obtained from adequate cost calculations to better assess feasibility.

6. Conclusions

MAR is one of the technically feasible options available to increase water resources and, although the economic analysis is not only the only variable involved in decision making, it is one of the most significant. The methods followed to analyse the effects of water resource projects are sometimes complex, may have to be applied by an interdisciplinary team and might not be well understood by non-specialists. Water resources management and the selection of the most suitable solution require providing enough accurate information to both government bodies and users, which means full cost estimations for each alternative. Most planning for water resources projects in the public sector involves having to consider both economic and non-economic objectives.

For water resources projects, the economic effect may involve beneficial or adverse effects, in addition to construction and operational costs and user benefits. Economic feasibility implies the discounted benefits of constructing and operating a project that exceeds the discounted costs throughout its life span; at a governmental level, the economic purpose is efficient public investment that focuses on net returns to society-at-large. However for the most suitable solution, a project must be justified 'on balance' by considering not only the financial and economic effects, but also the set of environmental, social, institutional, political impacts, and others (i.e. national efficiency, regional development and environmental quality). The fact that there are many engineering projects for which solutions are obvious, and for which detailed studies and alternatives are not undertaken, needs to be considered. For such projects, a sophisticated economic or financial study is not, therefore, required to meet clearly defined needs or legal requirements.

This application shows the interest in coupling results from groundwater modelling to assess *ex post* economic results in MAR. An *ex post* CBA has the advantage of being based on less speculative information since all costs and benefits have already occurred. However, it has less power to influence the resource allocations for the current project, but can affect resource allocations for similar future projects. Although determining benefits is troublesome, a more comprehensive economic analysis would consider the economic benefits of the industrial/agricultural benefits that arise for a more reliable water supply, saving water during droughts, etc. The project might also be expected to increase the labour and capital supplies in the project area, and to enable more intensive use of available land and water resources, as a result of increased tax revenue or foreign exchange earnings made from exporting crops. Other improvements in regional economy from the project implementation may also occur.

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