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Groundwater modeling to study brine disposal impact from desalination plant in Sharm El-Sheikh, South Sinai, Egypt

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ABSTRACT

Seawater desalination is considered a non-conventional source for suitable drinking water supply; still considerations about negatively environmental adverse effects are raised. Disposal of rejected brine represents major environmental challenges for most desalination plants. The aim of this paper is to explore the adverse environmental impacts of entailing huge amounts of brine into the local coastal aquifer along the Red Sea coast of Sharm El-Sheikh area, South Sinai. Brine water and ten production wells of the largest water desalination plant; El-Montazah, were sampled for all available chemical and trace elements analysis. The speciation and saturation with respect to minerals have been done using geochemical modeling indicating super-saturation of dolomite in feed water, while; calcite and dolomite were supersaturated minerals in brine water. Mathematical modeling of groundwater flow using MODFLOW-2000 was applied with variably-density miscible salt transport using SEAWAT code governing equations applying three different scenarios up to 50 y was used to investigate and predict the impact of brine disposal on the coastal aquifer. Simulation results indicate that; Miocene aquifer was affected by both sea water intrusions (laterally) and the effluent of brine water through the injection well (vertically). Some Mitigation actions were recommended to lessen the harmful and destructive environmental impacts of brine water disposal.

Keywords: Brine disposal; Environmental impact; Hydrochemistry; Groundwater modeling; Sharm El-Sheikh, Egypt

1. Introduction

One of the most universal challenges of the 21st century was water scarcity where, the worldwide current water resources are inadequate and depleting continuously due to population growth, rapid industrialization, overexploitation and climatic changes [1,2]. Seawater desalination is perceived as most viable and oldest process to encounter the mounting demand for high-quality water [3,4]. According to the International Desalination Association (IDA) report in 2019, there are 19,744 desalination plants worldwide, having a daily desalination capacity of around 99.7 million m^3 located in 150 different countries and serving the need of >300 million people [5]. Desalination water supply is about three-fold by 2050 [6]. Desalination technologies can be classified into thermal distillation (multi-stage flash (MSF), multi-effect distillation (MED) and membrane separation (RO), some plants use hybrid systems for desalination process. Reverse osmosis technique (RO) is considered a key technology in desalination processes. However, this technology holds severe environmental implications. One of these severe environmental impacts is the production of hypersaline brine water which is discharged to the nearshore environment. The brine is known as a concentrated or reject contains most of dissolved solids of the feed water in concentrated form and chemicals used

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for water pretreatment and post-treatment, such as acids, bases enzymes, antiscalants, biocides and detergents [7]. All these chemicals are added during desalination process and before brine disposal into the ocean that consequently enhance severe negative environmental impacts on the nearshore ecology. The brine volume production is estimated to be 141.5 million m^3/d that is discharged into ocean environment [8,9]. The repercussions of brine disposal on environment are categorized into direct and indirect impacts [10]; direct effects are related to brine-associated pollution (marine and groundwater pollution), air pollution due to greenhouse gases (GHGs) emissions and intensified use of energy. Whereas, indirect impacts are related to noise pollution, use of land and other social impacts. Aquifer/groundwater contamination including sea water temperature increment and lowering dissolved oxygen levels that have an effect on aquatic life, as well as, chlorination during water desalination process might introduce some toxic substances into water resources. Several studies were done for the adverse environmental impacts of desalination process [11–15]. Few studies predict the effects of brine disposal from desalination plants on coastal aquifers [16–19]. Some mitigation actions were taken to lessen these adverse environmental impacts of brine disposal through: (1) Source water intake by using low intake velocity and a combination of screen with different mesh sizes [20]. (2) Brine discharge method by releasing brine water through diffuser systems, thus eliminating the reliance of concentrate dilution on cooling water capacity and furthermore, the difference between the temperature of the reject stream and receiving water bodies can be minimized if cooling water is not employed for brine dilution [21,22]. (3) Zero-liquid discharge (ZLD) water desalination plants approaches [23,24]. Furthermore, the temperature, salinity and concentration of chemicals in the reject stream can be reduced by efficient brine dilution. It is essential to locate the discharge stream far away from productive and ecologically sensitive areas.

In Egypt, desalination has received increasing attention as a results of increasing demands for some activities (urbanization, tourist, and domestic) under constrain of natural fresh water resources scarcity at the coastal areas and limited rainfall recharge that doesn't exceed 200 mm/d at western and eastern Egyptian coastal area [25,26]. The Red Sea is considered to be one of the potential touristic zones facing water shortage crisis in Egypt, although any development strategies are primarily based on water availability. The Red Sea region has no surface water sources, low rainfall on highlands and high-saline coastal groundwater aquifers. A special focus on Sharm El-Sheikh as a major tourist city in south Sinai governorate has been done, where rainfall is scarce and groundwater was insufficient to meet water requirement for socio-economic development in the area. In addition to, Miocene aquifer (the main water bearing formation) possesses high salinity parameter and desalination process is considered the main source of freshwater for different purposes via construction of about 45 reverse osmosis (RO) plants [27,28].

El-Montazah Water Desalination Plant is an example for desalination plants that fed indirectly by seawater through the vertical wells intake (deep drilled Miocene groundwater wells) ranging in depths from 30 to 100 m as in (Fig. 1A). The production quantity of El-Montazah desalination plants reaches 18,000 m^3 in 2017 as in (Fig. 1B) [29]. In the RO desalination processes, the feed water are split into two solutions; the first is freshwater (30%–40%) and the second is the reject brine water (70%–60%, respectively). This brine contains most of the salts, dissolved minerals and chemical waste products. The most common method that has been used for the last decades in Sharm El-Sheikh for brine disposal is the injection well disposal method. In many cases, production wells are not far (less than 50 m) from brine disposal reject wells (over 60 m depth), consequently, raising the salinity of the intake groundwater source [30]. Although, in 1994, many verdicts were issued by the Egyptian Ministry of State for Environmental Affairs (EEAA) for forbidding brine disposal into the sea, but this law is not yet well enforced [31]. In Sharm El-Sheikh, groundwater deterioration is mainly attributed to high pumping rates under scarce annual precipitation, in addition to, deep disposal of brine from desalination plants at the sea coast, reject brine water contains concentrated chemicals that cause damage and reef degradation in the

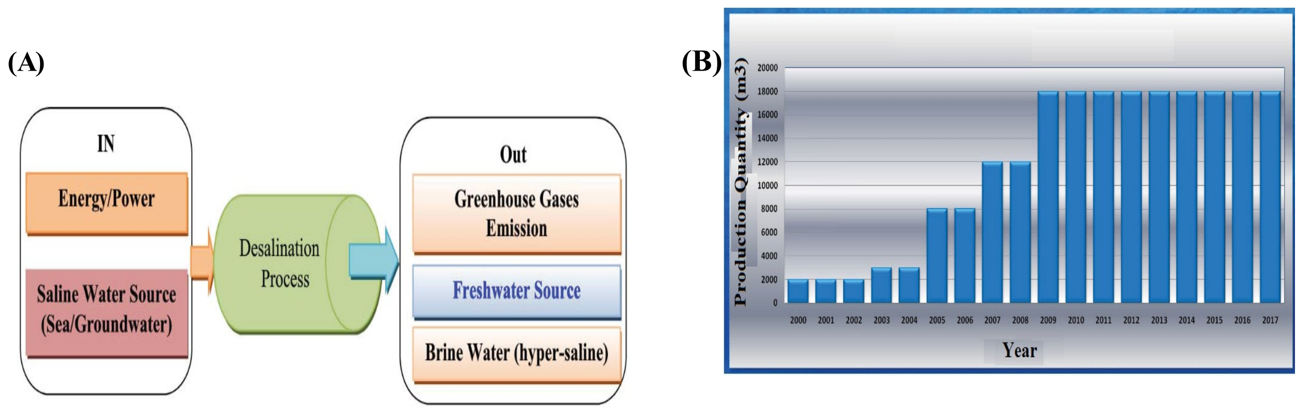

Fig. 1. (A) Schematic diagram for desalination process and (B) production quantity in (m^3)/y from El-Montazah Water Desalination Company.

National Park area in Sharm El-Sheikh area [32]. In the last century, South Sinai especially, Sharm El-Sheikh has experienced a significant increase in desalinated water production to fill the gap due to the continuous water demand. The capacity of desalination plants in Sharm El-Sheikh has increased from 25,000 m^3/d in 2001 [33] to 150,000 m^3/d in 2018; entailing large amounts of desalination reject (brine) into the local aquifers. This has caused an increase in the salinity of the feed water from 44,000 mg/L in 2001 to 55,000 mg/L in 2018 at specific locations. In addition to, the environmental degradation in Sharm El-Sheikh environs due to the bloom in desalination and use of chemicals in more than one stage during desalination process. This increased the total cost of water desalination and threatens the sustainable development.

The aim of this paper is to investigate the negative environmental effect of brine disposal on coastal groundwater aquifer in Sharm El-Sheikh area. This will be implemented through the exploration of hydrogeochemical processes that affect groundwater conditions in conjugation with simulation of current status of groundwater flow and variably density solute transport using MODFLOW-2000 and SEAWAT code with special focusing on some aquifer salinization mitigation strategic plans. Several scenarios of pumping rates and injection have been analyzed to address the effects of anticipated future demands on the hydrogeological system in the study area.

2. Site description

Sharm El-Sheikh is extends as an elongated narrow coastal strip on the southwestern coast of the Gulf of Aqaba, on the extreme southern tip of Sinai Peninsula the study area is located between longitudes 34° 12′ and 34° 18′ E and latitudes 27° 40′ and 28° 00′ N (Fig. 3). The climate in this region is extremely arid with a rare rainfall averaging 25 mm/y and thunder showers causing occasional intense flooding events. The average monthly daily temperature ranges between 17°C in January and 32°C in August, while the relative humidity ranges between 32% in July and 48% in December [34]. In general, annual precipitation and flood storms represent the main sources of recharge for shallow groundwater aquifers in Gulf of Aqaba drainage basin system [35]. The total annual groundwater recharge ranges between 5% and 20% of total annual precipitations and flood storms [36] which occurs in October and January each year [37,38]. Sharm El-Sheikh area represents the outlet of drainage basins extending northward, where the drainage stream stem encountered the Miocene aquifer. In 1997, the static groundwater level ranged from 4.63 to 5.7 m (amsl) [39]. The static groundwater depth declined from 42 to 75 m between July 1996 and March 2007 [40]. The average groundwater salinity showed an increase during the last decades: 27,550 mg/L in 1997; 33,000 mg/L in 2002; 34,000 mg/L in 2007 [41]; 56,443 mg/L in 2013 [42] and finally, 40,300.8 mg/L in the present study. The data scarcity of this area is challenging problem for groundwater investigations at practical scales, hence most studies are sporadic and localized.

The main water bearing formation in the study area is Miocene aquifer in addition to, the Quaternary alluvial sediments and the fractured basement [45]. The Miocene aquifer is mainly composed of limestone, dolomitic limestone, and coarse grained sandstone with shale intercalations, and the Quaternary aquifer which is composed of gravel, sand and silt. The seasonal rainfall and flash floods can be considered as the main sources for recharging both of Quaternary and Miocene aquifer; although the Quaternary aquifer has high recharge opportunity from precipitation and surface runoff, it is not explored yet in the study area.

3. Materials and methods

3.1. Data collection

A set of representative 10 groundwater samples were collected and analyzed for physical measurements on in-situ field measurement including electrical conductivity (EC), pH, temperature, total dissolved solids (TDS) using portable probes. For chemical parameters comprising major cations (Ca, Mg, Na, K) and anions (Cl, SO_4, HCO_3) concentrations. All the wells located in the studied area had been in use during the time of sampling (Fig. 2A). The samples were collected in 1-L narrow neck pre–washed polyethylene bottles. Analysis of the water samples was carried out following the methods described in ASTM [46]. Total hardness (TH) as $CaCO_3$ and Ca^{2+} were analyzed using standard EDTA. Mg^{2+} was calculated by taking the differential value between TH and Ca^{2+} concentrations. Na^+ and K^+ were measured using a flame photometer. Total alkalinity and $CaCO_3$, CO_3^{2-}, and HCO_3^- were estimated by titrating with HCl. Cl^- was determined by standard Hg $(NO_3)_2$ titration. SO_4^{2-} was analyzed using UV/visible spectrophotometer. All parameters are expressed in milligrams per liter and milliequivalents per liter. Data quality was assessed using the charge balance between the difference of cations and anions (expressed in meq/l) divided by their summation according to the following equation:

$$\frac{\sum(\text{Cations} - \text{Anions})}{\sum(\text{Cations} + \text{Anions})} \times 100 \quad (1)$$

With an acceptable range of ±5 [47] that confirm the water quality assessment. Heavy metals were measured using Thermo ICP-OES (inductively coupled plasma-optical emission spectroscopy) were achieved in the central lab of Desert Research Center. All reported values have an ionic balance within 5% all concentrations were reported in mg/L.

3.2. Hydro-chemical modeling

PHREEQC Interactive, Version 3.3.9 software (the Lawrence Livermore National Laboratory Model and WATEQ4F) was used to carry out the hydrochemical modeling for speciation and saturation index calculations with respect to minerals and gaseous phases of analyzed samples from groundwater and brine water to characterize the actual state of the hydro-chemical system in the study area. As a results of partially high salinization of the solutions with ionic strengths up to >1 mol/L, a

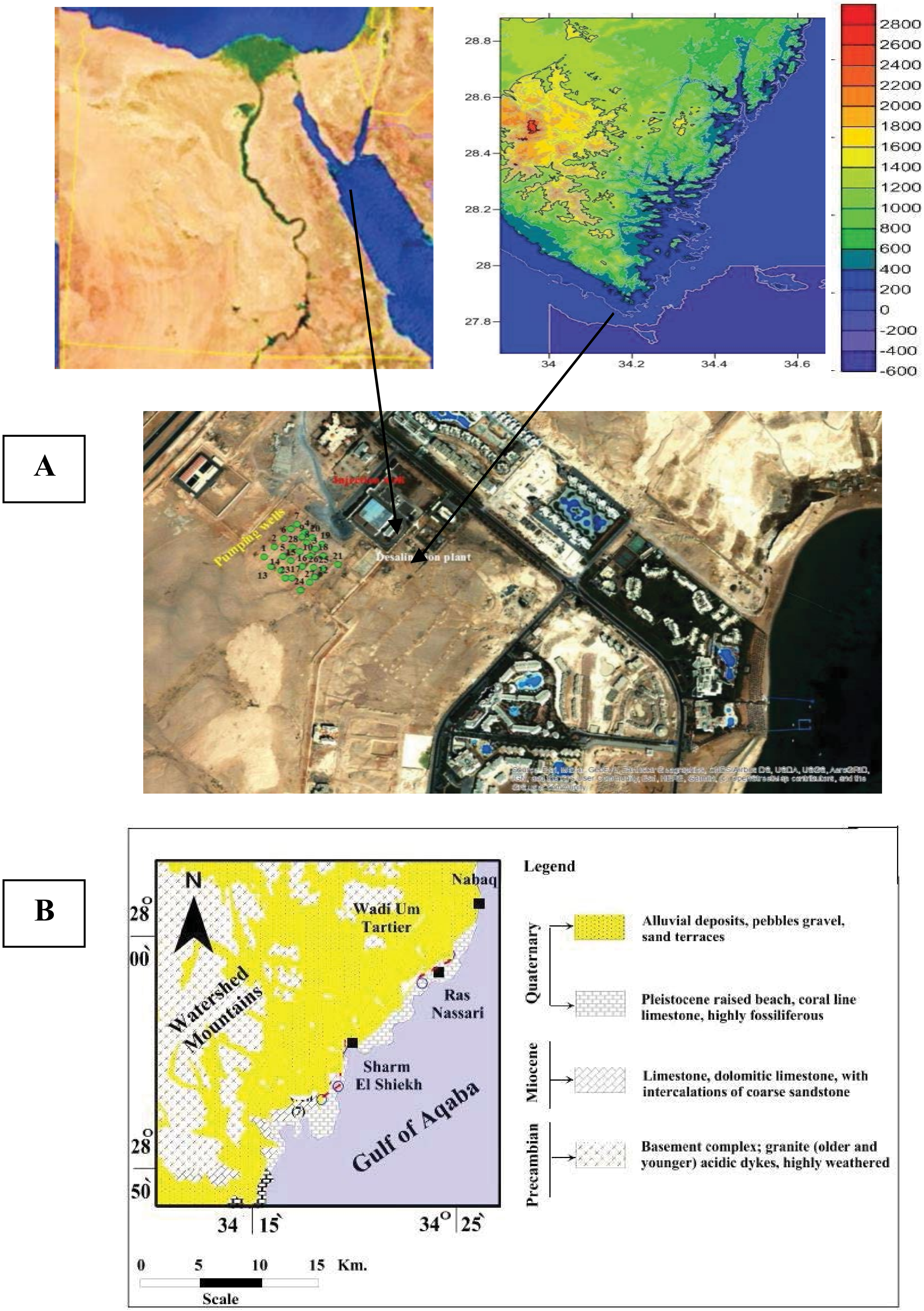

Fig. 2. Site description: (A) location map of collected water samples and (B) geological map of the study area (modified after [43,44]).

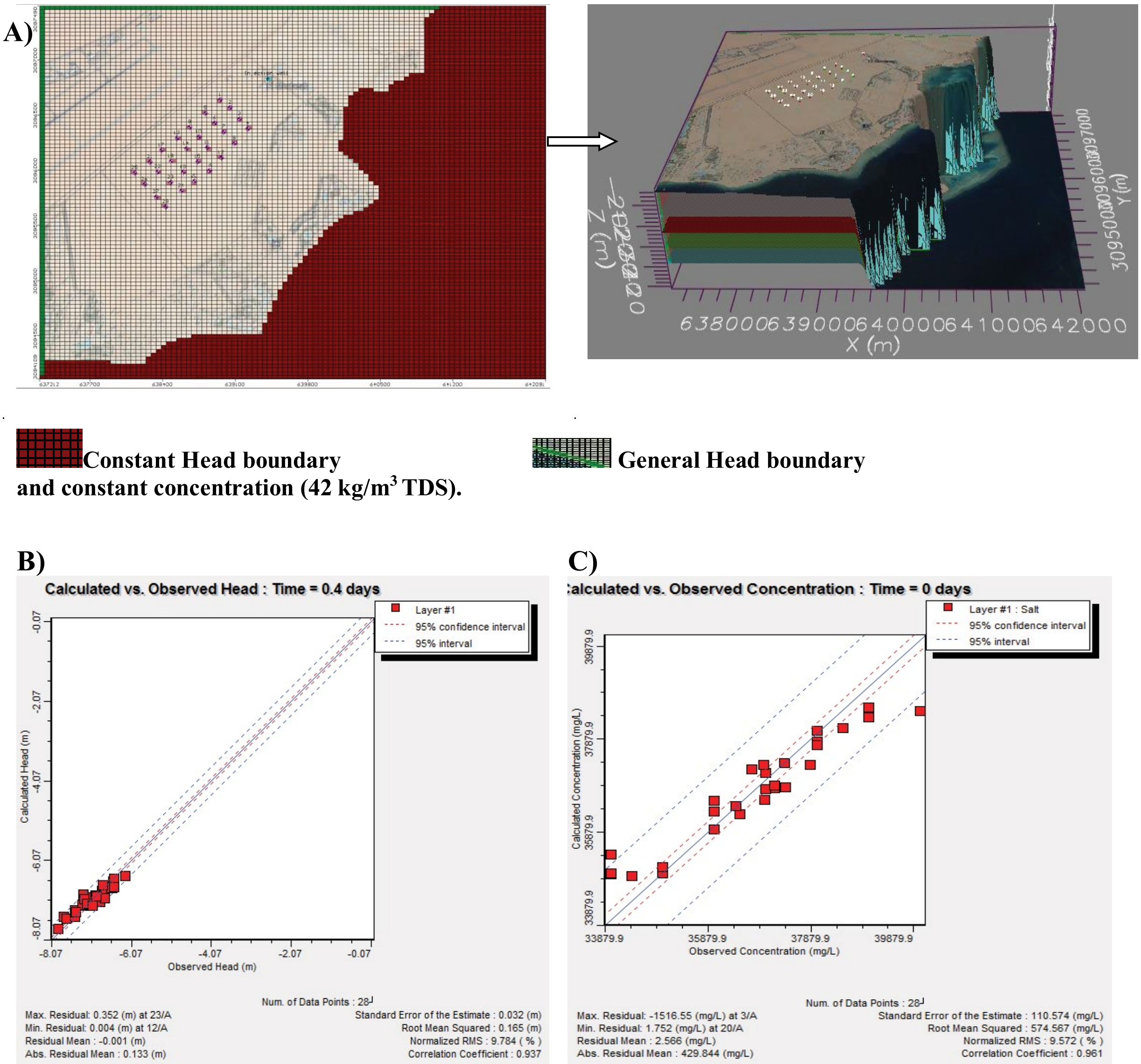

Fig. 3. (A) Finite-difference grid and boundary conditions for Sharm El-Sheikh shallow aquifer. (B) Calibration of observed and calculated heads in 2020. (C) Calculated vs. observed groundwater salinity in 2020.

thermodynamic database basing on solubility data which were first developed for the complete system parameter including all analyzed hydrochemical parameters at 25°C and 1 atm pressure based on the compilations of [48,49] (included as the parameter file "Pitzer.dat" in PHREEQC Interactive, Version 3.3.9 2016).

3.3. Groundwater flow modeling and forecasting of seawater intrusion processes

Integration of groundwater flow and solute transport computations allow to study the effects of fluid density gradients associated with solute concentration gradients to be incorporated into groundwater flow simulations (i.e., density dependent flow). In this work; the finite difference method of numerical integration have been used to solve 3-D confined and unconfined groundwater flow under many types of natural and artificial aquifer stresses in shallow aquifer of Sharm El-Sheikh area. The flow model was constructed using SEAWAT-2000 a computer program that combines a modified version of MODFLOW-2000 with MT3DMS [50,51]. An implicit finite difference (FD) method was selected to solve the advection equation and central-in-space weighting was specified to minimize numerical dispersion.

3.3.1. Spatial and temporal model discretization

The model domain and finite difference grid used to simulate groundwater flow within Sharm El-Sheikh coastal aquifer is illustrated in Fig. 3A. The grid consists of 100 rows and 100 columns in plain view. Each cell is 48.79 × 33.81 m² in the horizontal plane. Coastal unconfined Miocene aquifer was treated as a single layer with a mean hydraulic conductivity equal 10–50 m/d [52]. Longitudinal and lateral dispersivities were set equal to 10 m and 1 m respectively. The effective porosity was set equal to 35%.The eastern boundary is saline seawater transport boundary (41.4 g/L) [53] since the Miocene aquifer was in direct hydraulic contact with the Gulf of Aqaba in which the water level is known (zero). At the western and northern boundaries general head boundaries conditions were applied, for the upper boundary, the water table was considered lower than the Mediterranean Seawater level. The concentration was constant and equal to the groundwater concentration. The bottom boundary was impermeable, that is, the normal flux through the bed for both fluid and salts was equal to zero. The input data required for the model include: well location and pumping rates, aquifer configuration, including top and bottom of the system; hydraulic conductivity of the system; amount of recharge and potentiometric head.

3.3.2. Model calibration

Model calibration was achieved through several trials and errors by adjusting the values of recharge at the boundaries. The calibration has been conducted vs. potentiometric head data at 2020 to calibrate the spatially variable hydraulic conductivity and recharge, and hydraulic conductivity of the general head boundary. Calibration process produced an acceptable comparison between observed vs. calibrated heads and concentration in mg/L (Fig. 3B and C).

3.3.3. Testing scenarios

Three proposed scenarios were simulated after model calibration to study brine disposal effects on groundwater system.

- The first considered pumping rate of 120 m^3/h only for 10 wells as the rest are turned off (base case scenario).
- The second scenario considered the same pumping rate of 120 m^3/h for all pumping wells at the same time.
- The third scenario considered elimination of injection well or turned off.

4. Results and discussion

4.1. Environmental impacts of brine disposal

4.1.1. Physicochemical impacts

The results of chemical analysis of selected feed water for El-Montazah water desalination station and the injection well are summarized in Table.1. The feed water wells tapping Miocene aquifer have TDS values range from 40.26–40.32 g/L. The concentrations of both cations and anions in the feeding water group are quite similar, the variation in all species are nearly limited. On the other hand, TDS of the injected brine sample was recorded to be 51.95 g/L; which is considered to be higher than Aqaba Gulf water (41.6 g/L) by 1.24 fold. The injected brine water might introduce higher salinity pollution risk to the surrounding groundwater system rather than the induced seawater intrusion.

4.1.2. Chlorine concentration

In RO plants, the intake water is chlorinated to reduce biofouling, but dechlorinated again with sodium bisulfite before the water enters the RO units to prevent membrane damage. This step might increase the concentration of residual chlorine that may influence the water quality of the ambient water and ecological system. The concentration of Cl^- in the brine of the study area is 29.4 g/L which is greater than the feeding water (average concentration of Cl^- is 19.6 g/L) revealing that the chlorine in the brine has increased by 1.5 fold.

4.1.3. Heavy metals

Trace element analysis of feed water and brine water are shown in Table 2 contain traces of iron, nickel, chromium and molybdenum; these metals may be result from corrosion of various equipment although the desalination plant is using the non-metal equipment and stainless steels.

Table 1
Chemical analysis of selected feed water and injection well

Type	Well	pH	EC	TDS	Cations concentration (mg/L)				Anions concentration (mg/L)		
			μs/cm	mg/L	Ca	Mg	Na	K	HCO_3	SO_4	Cl
	1	7.5	62,950	40,288	574.11	614.11	11,000	3,320	61	4,400	20,063
	2	7.5	62,900	40,256	574.1	614.1	12,000	330	61	3,860	19,329
	3	7.5	63,000	40,320	574.9	614.9	12,200	330	67.1	2,600	20,063
	4	7.5	63,000	40,320	574.8	614.8	12,400	330	67.1	3,862.2	19,574
Feed	5	7.5	62,950	40,288	574.17	614.17	12,200	320	67.1	3,860	19,574
water	6	7.5	63,000	40,320	574.12	614.12	12,200	330	67.1	3,850	19,574
	7	7.5	63,000	40,320	574.13	614.13	12,000	320	61	3,860	19,574
	8	7.5	62,900	40,256	574.14	614.14	12,400	320	61	3,860	19,574
	9	7.5	63,000	40,320	574.15	614.15	12,400	330	67.1	3,850	20,063
	10	7.5	63,000	40,320	574.16	614.16	12,000	320	67.1	3,840	19,574
Injection well		7.4	88,500	51,949	712.4	710.3	17,500	510	109.8	3,100	29,361

Table 2
Trace element results of some selected feed water and brine

Item (mg/L)	1	2	3	4	5	Injection well
Al	0.147	0.1106	0.1203	0.1433	0.1196	0.121
B	9.203	6.208	8.716	5.817	1.059	7.807
Ba	0.0712	0.0267	0.0306	0.0186	0.0566	0.0361
Cd	0.016	0.0099	0.0145	0.0120	0.0229	0.0063
Co	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Cr	0.0339	<0.01	<0.01	<0.01	0.0361	<0.01
Cu	0.0452	<0.006	0.0518	0.0373	0.0121	0.0718
Fe	<0.02	0.057	0.1369	0.1325	1.886	0.1613
Mn	<0.002	<0.002	<0.002	<0.002	0.0732	<0.002
Mo	0.2233	0.0715	0.0689	0.0837	0.0227	0.225
Ni	<0.002	<0.002	0.0167	<0.002	<0.002	<0.002
Pb	<0.008	<0.008	<0.008	<0.008	<0.008	<0.008
Si	9.929	3.258	5.068	5.020	8.910	7.689
Sr	26.16	13.65	20.14	12.14	6.721	17.64
V	0.0517	0.0604	<0.01	0.0418	<0.01	0.0351
Zn	0.0361	0.024	0.0204	0.0105	0.0193	0.0224

The presence of these metals is of concern if present in high concentrations which consequently not safe for the aquatic life.

4.2. Geochemical modeling (phase equilibria)

Saturation Index is vital geochemical parameter in the fields of hydrogeology and geochemistry, often useful for identifying the existence of some common minerals in the groundwater system [54,55]. In this present study, saturation indices (SIs) with respect to gypsum, anhydrite, calcite, dolomite, aragonite and halite were calculated using PHREEQC. Saturation index results in Table 3 indicated that feed water samples were supersaturated with dolomite $CaMg(CO_3)_2$ and undersaturated with calcite, anhydrite, gypsum and halite minerals; widely accepted hypothesis of dolomitization is that limestone is transformed into dolomite by the dissolution of calcite followed by dolomite precipitation. For brine water, calcite and dolomite were supersaturated minerals as a result of the anti-scaling chemicals used to prevent calcite precipitation.

4.3. Model simulation results

4.3.1. Flow pattern with brine injection

The model results give the contours for the equipotential head, drawdown, velocity and the water table in the simulated aquifer system with TDS values vs. time for monitoring wells located at Sharm El-Sheikh area.

Table 3
Saturation index results of feed water and brine water

Well	Calcite	Dolomite	Anhydrite	Gypsum	Halite
	$CaCO_3$	$CaMg(CO_3)_2$	$CaSO_4$	$CaSO_4$	NaCl
3	–0.121	1.154	–0.714	–0.441	–2.567
4	–0.115	1.166	–0.764	–0.491	–2.541
5	–0.063	1.277	–0.93	–0.657	–2.513
6	–0.076	1.246	–0.772	–0.499	–2.522
9	–0.075	1.247	–0.769	–0.496	–2.529
12	–0.121	1.154	–0.714	–0.441	–2.567
15	–0.115	1.166	–0.764	–0.491	–2.541
20	–0.063	1.277	–0.93	–0.657	–2.513
22	–0.076	1.246	–0.772	–0.499	–2.522
28	–0.075	1.247	–0.769	–0.496	–2.529
Injection well	0.099	1.571	–0.888	–0.623	–2.211

According to Fig. 4 for head evolution in the study area, it was noticed that groundwater flow was directed from northeastern to southwestern direction that enhance seawater intrusion with salinity increments along the simulation time. As brine injection begins it will change the hydrological pattern and spreads in all directions vertically and laterally creating a high salinity and density plume. The water table level was fluctuated due to the effect of proposed pumping scenarios (stresses on the aquifer system) with flow velocity of 0.72 m/d across the model area. Furthermore, this flow velocity within the brine plume is higher than the flow on the offshore side.

4.3.2. Combined effect of seawater intrusion and brine injection on aquifer system

The simulated results of variable density salt in mg/L are plotted verses time for pumping wells located at Sharm El-Sheikh area are represented in Fig. 5A–C for the three proposed scenarios several observations will be described as follow:

- After simulating brine injection for 50 y as in 2D-salinity simulation scenario (Fig. 5A), saltwater inflow rate increases depending on the rate of injection and injection location.
- For 3D-salinity simulation evolution in scenario 2 (Fig. 5B), it was noticed that salinity evolution started to increase vertically through the aquifer layers depending on rate of pumping from the operating wells and time of operation of these wells.
- Although El-Montazah Water Desalination Plant was inland (far about ~1 km from the coast), Figs. 5 shows that continued pumping increased salinity plume evolution through Miocene aquifer layers (TDS = 41 g/L) in about 50 y from 2020 as a result of mixing of high density brine water discharging into saline aquifer via injection well in addition to seawater intrusion on the long time in case of all pumping wells were operated in scenario 2 (Fig. 5B) than scenario 1 (Fig. 5A) where only ten pumping wells were operated.
- When injection well was turned off /eliminated as shown in (Fig. 5C), the effect of seawater intrusion was noticed only with 10 pumping wells were operated.
- Groundwater of Miocene aquifer in Sharm El-Sheikh was affected by both seawater intrusions (laterally) and injection of brine water into the aquifer by direct/indirect methods, consequently, doubled or tripled salinity values on the long time.
- Also, decrease in water table (m) with increment in salinization of some wells might be due to over-pumping that may led to dissolution of some marine deposits an additional factor that led to additional increment in TDS (mg/L) values.

5. Conclusion and recommendation

The objective of this study was to evaluate the negatively environmental impact of brine water disposal on Miocene groundwater coastal aquifer of Sharm El-Sheikh area, south Sinai, Egypt. An integration of hydrochemistry with groundwater flow and transport numerical model for variably density salt (MODFLOW-SEAWAT) model was applied to solve finite difference equations of variable density system for 50 y of simulation. This was achieved through collection of about 10 feed groundwater samples for El-Montazah Water Desalination Plant and carrying out physical and chemical analysis of these samples in addition to one brine water sample.

Results of model simulation indicated that Miocene coastal aquifer is affected by both lateral seawater intrusion and the effluent of injected brine water through aquifer layers. The impact of seawater intrusion on the groundwater increases with decreasing distance from the coast where, El-Montazah Water Desalination Plant was far about 1 km from the coast. Seawater intrusion is forced by the continuous abstraction of the non-renewable brackish groundwater, which is a problem for the region due to its arid climate and negligible actual groundwater recharge. Consequently, the regions of brine infiltration and injection are not only characterized by increased salinity but also by saturation and supersaturation with respect to dolomite and calcite minerals. As a result, an intense dolomite precipitation in fact occurs in the screens of wells extracting high concentrated water, requiring a frequent wash over. The effluent of brine water into groundwater systems causes harmful and destructive environmental impacts (physiochemical and ecological impacts). As Egyptian

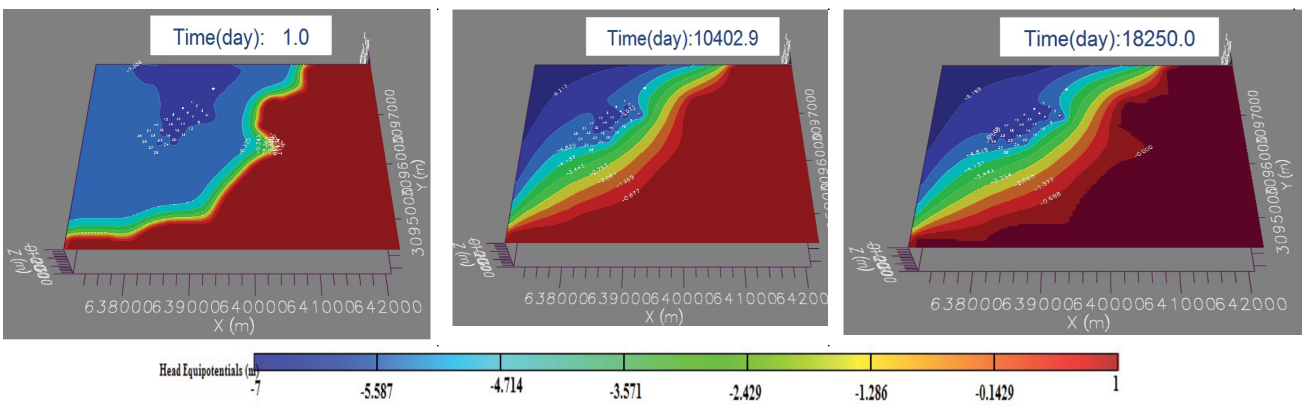

Fig. 4. Head evolution through Miocene aquifer layers in Sharm El-Sheikh area (base case scenario).

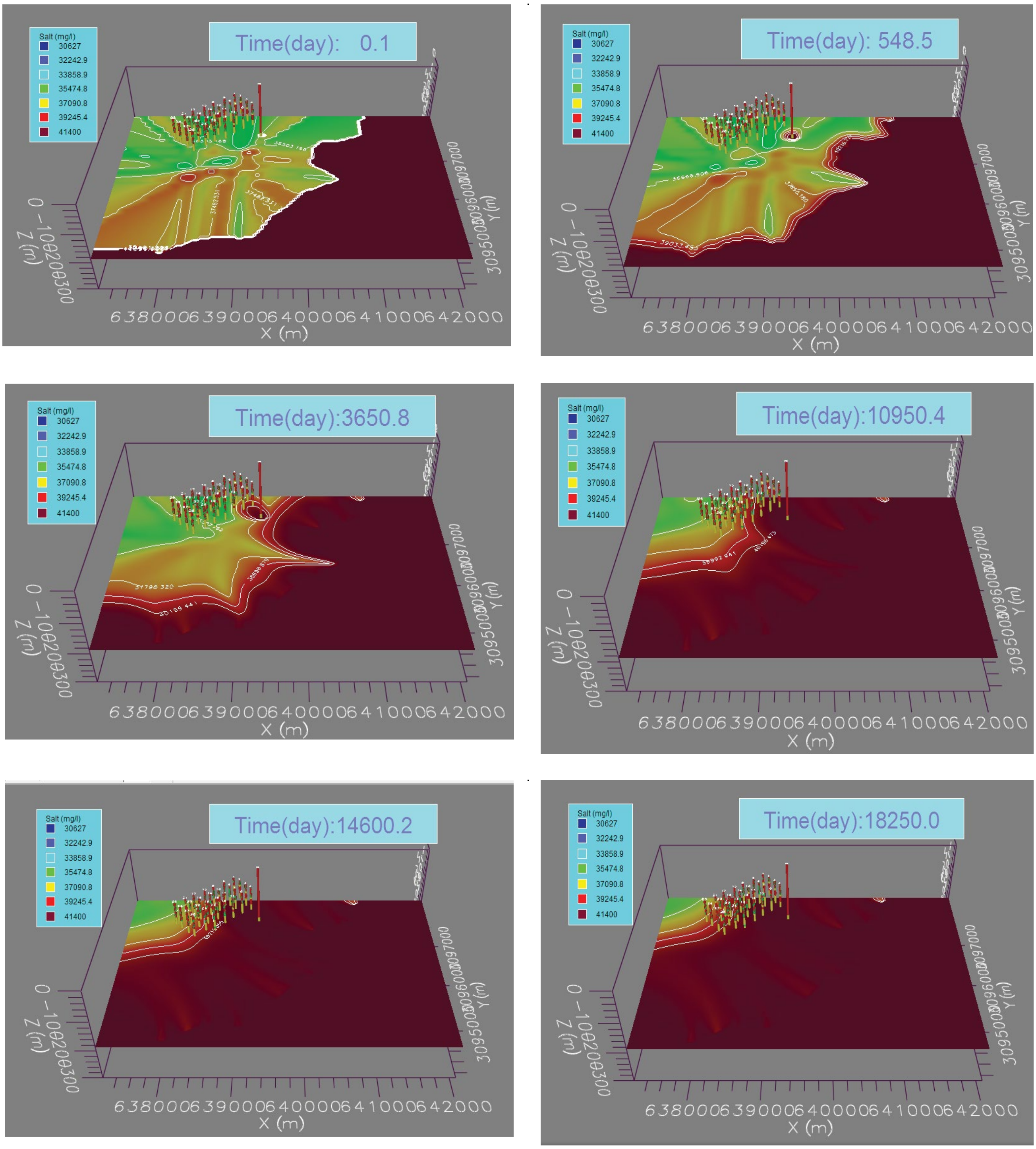

Fig. 5. (A) 2D-salinity plume evolution through Miocene aquifer layers in Sharm El-Sheikh area (base case scenario).

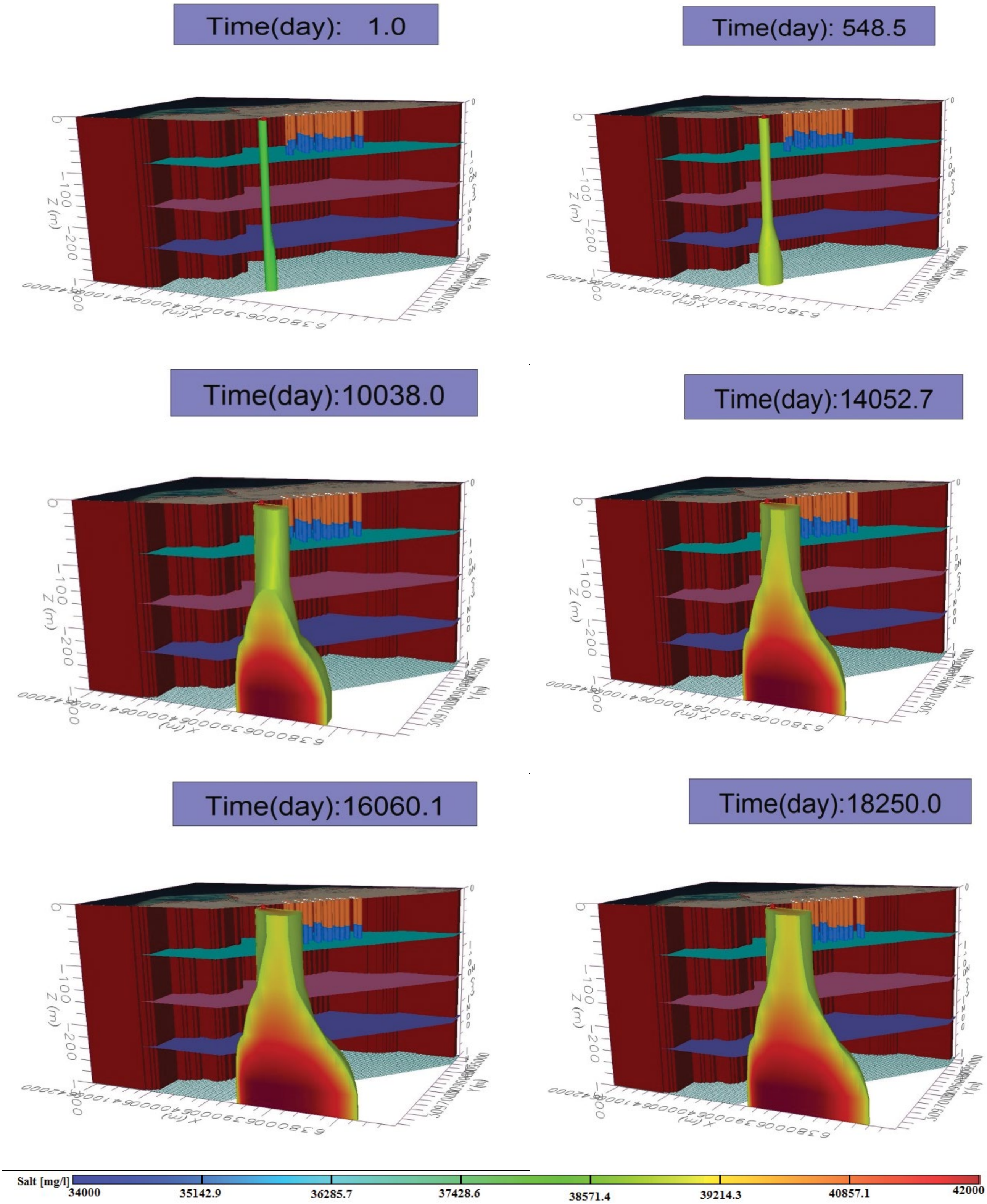

Fig. 5. (B) 3D-salinity plume evolution through Miocene aquifer layers in Sharm El-Sheikh area (proposed scenario 1).

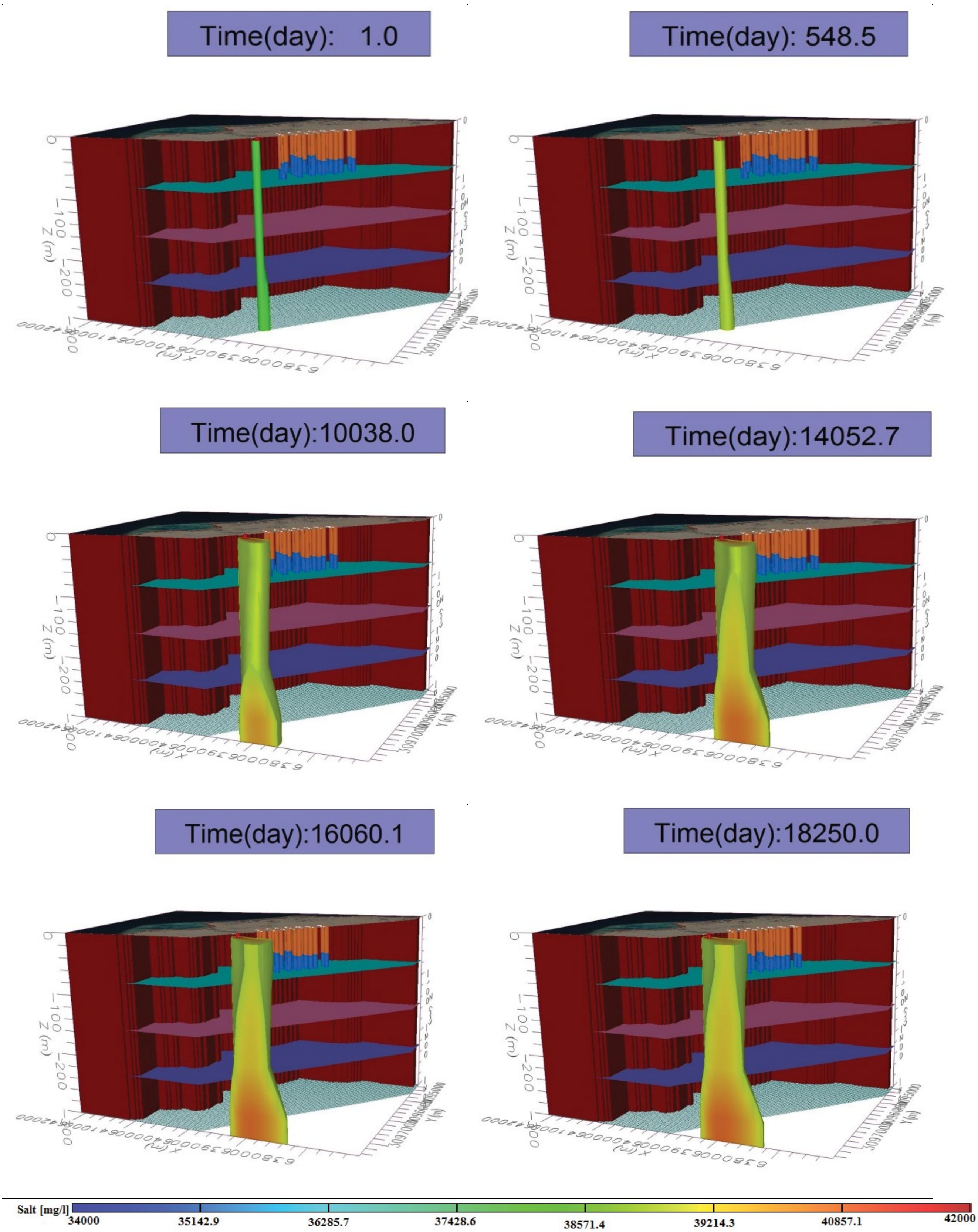

Fig. 5. (C) 3D-salinity plume evolution through Miocene aquifer layers in Sharm El-Sheikh area (proposed scenario 2).

Ministry of State for Environmental Affairs (EEAA) issued many verdicts in 1994 for brine disposal forbidden into the sea, but this law is not yet well enforced [56]. So, attempts have to be done to find some mitigation actions to reduce these harmful environmental impacts of brine disposal in coastal aquifers. The brine injection into the saline part of the coastal aquifer could be recommended as a mitigation action or even through the aquifer deepest parts where the flow was reversed.

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