



# Groundwater monitoring for assessing artificial recharge in the Mediterranean coastal aquifer of Korba (Northeastern Tunisia)

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**Abstract** Groundwater (GW) artificial recharge with treated wastewater (TWW) is seen as a promising technical solution to overcome groundwater overexploitation and seawater intrusion and to ascertain future water resources availability. While these introduced recharge systems have shown positive impacts in many areas across the world, their effective contribution to GW recharge is still not well quantified in water-stressed regions such as Tunisia. The scarcity of on-site data with accurate temporal and spatial measurements and resolution in such areas hinders the clear assessment of these recharge systems' performances. This article presents an approach based on the on-site monitoring of groundwater level and salinity measurements and recharge rate records spanning over the 6-year period to investigate the performances of a wastewater-based recharge system located in the Korba-El Mida site in Tunisia. The analysis showed that the artificial recharge did not have an impact on groundwater level.

Controversially, a decrease in water salinity in and around the study site was observed. Findings emphasize on the weak wastewater volumes applied to the recharge system and the clogging phenomena that occurred in the pool and in the vadose zone. Furthermore, analyzing the information over a short period of time could be misleading as the results of a 6-year period of monitoring show clearly that the way the site was managed and the local conditions have a role in understanding and interpreting the results even if the artificial recharge was appropriately designed.

**Keywords** Groundwater · Artificial recharge · Treated wastewater · Monitoring

## Introduction

Artificial recharge is a method to balance and recover groundwater (GW) resources through floodwater spreading systems and injection wells (Bouwer 2002). It has already proved its efficiency on groundwater level recovering in many aquifers under various environmental and socioeconomical settings across the world (Bekele et al. 2011; Hashemi et al. 2013). GW artificial recharge with conventional and nonconventional water resources is widely seen as a promising technical solution to overcome groundwater overexploitation and seawater intrusion and to identify future water resources availability (Sheng 2005; Cazorra 2008; Shammass 2008; Al-Assa'd and Abdulla 2010; Valhondo et al. 2020). In many coastal aquifers, GW artificial recharge

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systems are applied against seawater intrusion, but their efficiency and reliability are difficult to assess not only because of the complexity of the GW systems but also because of human-made various activities on GW system such as excessive water pumping and extraction from wells, agricultural and drainage channels, or rapid urbanization (Milnes and Renard 2004; Huang et al. 2013; Bedbabis et al. 2015; Stone et al. 2019).

In Tunisia, coastal aquifers are under pressures where overexploitation of GW resources, mainly for agriculture purposes, has exacerbated threats under different forms of pollution, GW level depletion, salinization, and seawater intrusion (Ben Hamouda et al. 2011; El Ayni et al. 2013). GW artificial recharge in Tunisia has started in the 1970s as experimental systems and was effectively used in water management since the earliest 1990s. The technique has shown good results to recover GW levels. For instance, the Teboulba coastal aquifer located on the coast of Tunisia witnessed an increase of 30 m in groundwater levels (Bouri and Ben Dhia 2010). The Kairouan aquifer in central Tunisia showed a significant rise of water levels between 0.2 and 5.25 m in 5 months of artificial recharge using a total water volume of 13 Mm<sup>3</sup> released from the Sidi Saad dam (Bel Hadj Salem et al. 2012). The authors reported also that the recharge has impacted in a radius of 8 km surrounding the injection wells by increasing the water level. Ketata et al. (2014) confirmed that the increase of water levels of El Khairat underflow aquifer reached 2.6 m following the recharge in the wadi from 2002 to 2005. However, Tunisia recognized the need to adapt GW artificial recharge using the treated wastewater (TWW) as an alternative to conventional water resources, yet it is limited and vulnerable. It has started in the middle of the 1980s for irrigation purposes and against saltwater intrusion (Neubert and Benabdallah 2003; Ben Brahim-Neji et al. 2014). However, only few studies have focused on assessing the performances of these introduced recharge systems. For instance, Jellali et al. (2010) studied the risk of wastewater on the aquifer using the experimental laboratory columns and the Hydrus-1D model. They concluded that nitrogen ammonia (N-NH<sub>4</sub><sup>+</sup>) attenuation in the groundwater could be ensured by an appropriate management of the recharge process in terms of recharge duration and rest period. Cary et al. (2013) used Boron isotopes and the carbamazepine (CBZ) to study the groundwater quality and the effect of the artificial recharge with TWW in the Korba aquifer in northeast Tunisia. They confirmed that the aquifer is affected by

seawater intrusion despite the meteoric recharge along the coastal line. Others' studies within the same aquifer reported controversial results regarding the performances of the artificial recharge system. For instance, Gaaloul et al. (2012) considered that the artificial recharge has induced a rise of the water levels causing a reduction of the vadose zone thickness from 15 to less than 4.5 m based on a nearby piezometer. Ouelhazi et al. (2014) assessed an increase of the groundwater level exceeding 1.5 m and an improvement of the groundwater quality of the Korba aquifer following 4 years of observations. In many cases, the assessment was based only on few local observations or limited on-site measurements of water level in these areas.

Indeed, the quantification of GW recharge is difficult because it cannot be measured directly with the relevant temporal and spatial scale, and a successful assessment of the performance and reliability of these introduced systems relies on the availability of accurate on-site physical data with adequate temporal and spatial resolution including information on historic piezometric variability, pumping and extracted water volumes, water quality, etc. which are not usually available (Singh et al. 2019). An on-site monitoring is often used to assess GW recharge systems and the impacts of the environmental changes on the spatiotemporal quantitative and qualitative groundwater variability. A comprehensive assessment based on accurate on-site measurements and data collection can lead to a successful identification of threats and impacts on groundwater system and, therefore, suggests adaptive remediation strategy (Horriche and Besbes 2006; El Yaouti et al. 2009; Ibrakhimov et al. 2011; Chaudhuri and Ale 2014).

Therefore, in this article, we assess the performances and reliability of the artificial recharge of GW system of the Korba region using the TWW based on an intensive on-site monitoring of water level coupled with local measurements and historic records over 6 years of water salinity spatially distributed over the Korba aquifer.

## Material and methods

The Korba aquifer is located in northeast Tunisia in the Cap Bon peninsula (Fig. 1 a and b). It is limited by the Abderrahmane mountain in the northwest and by the Mediterranean Sea in the southeast. The region is characterized by a semiarid climate with an average annual rainfall of 450 mm and annual temperature of 19.7 °C.

The region is known by its important activities of agriculture, food industry, and tourism (Ben Hamouda et al. 2011; El Ayni et al. 2013). Artificial recharge with freshwater transferred from Medjerda watershed to Cap Bon channel and from in situ dams is already applied in Korba, Diar Elhojjej, Tafeloune, and Lebna since 1999 through injection wells (Fig. 1c). Infiltration from irrigation return flow is also a source of the aquifer recharge.

The shallow aquifer is contained in the Plio-Quaternary deposits which are characterized mainly by sand and alternating thin layers of clays and sandstone (Fig. 2). Bellali et al. (2018) showed heterogeneous multilayered aquifers of Oligo-Mio-Plio-Quaternary sediments formed by the arrangement of sand, sandstone, and limestone bodies reaching over a 3000-m depth, separated by impermeable clay packages. The Plio-Quaternary reservoir corresponds to the shallow aquifer which is assumed to be impacted by the artificial recharge in Korba-El Mida site. The average transmissivity and hydraulic conductivity of this reservoir are, respectively, equal to  $2.5 \cdot 10^{-3} \text{ m}^2/\text{s}$  and  $1.5 \cdot 10^{-5} \text{ m/s}$  (Bellali et al. 2018). Nearby the artificial recharge site, transmissivity is measured to  $4 \cdot 10^{-3} \text{ m}^2/\text{s}$ , and the hydraulic conductivity ranges between  $2 \cdot 10^{-3}$  and  $2 \cdot 10^{-6} \text{ m/s}$  for, respectively, fine sands and sandstone (Jarraya Horriche and Benabdallah 2020). The storage coefficient is about  $5 \cdot 10^{-4}$ .

Aquifer water extraction is mainly performed by shallow wells for irrigation purposes. The annual extracted water volume was estimated to be 60 MCM in 2015 throughout 9500 wells, inducing an overexploitation of the aquifer. This situation is confirmed mainly in the areas of Korba, Diar El Hojjej, and Tafeloune (Fig. 1), where the maximum decrease of groundwater level was reported (Ennabli 1980; Paniconi et al. 2001; Kerrou et al. 2010). In fact, piezometric maps indicated a general flow from the northwest to the southeast during the 1960s, and the water salinity was lower than 3 g/l during this period. Since the 1970s, groundwater levels have been declining, resulting in an inversion of the hydraulic gradient and degradation of groundwater quality due to the combined effects of overexploitation of the aquifer water resources and to seawater intrusion (Kouzana et al. 2009; Slama and Bouhlila 2017). In addition, groundwater salinity was accentuated as responses to the use of fertilizers in the irrigated areas within the catchment. Ben Hamouda et al. (2011) confirmed that groundwater sampling in wells located near

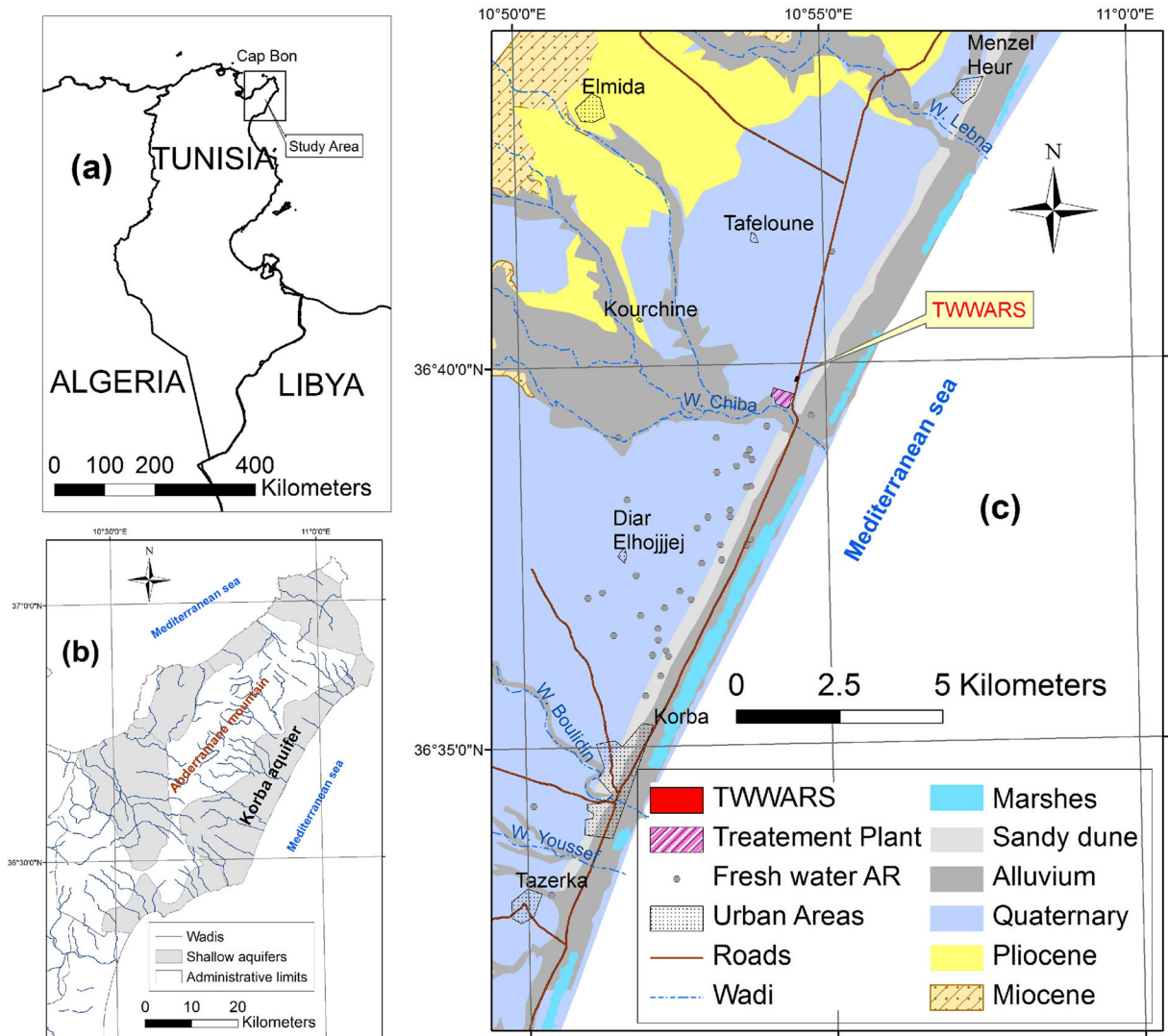
the irrigated areas and far from the coastal area was characterized by high nitrate content and chlorinated and sulfated chemical trend, whereas those located near the coast were characterized by Na-Cl trend. Thus, they concluded that groundwater salinization was due to seawater intrusion near the coast, whereas the contamination of wells far from the coast was related to the irrigation return flow. El Ayni et al. (2013) endorsed these findings based on chemical, physicochemical, and microbiological analyses. The authors reported also the need for further adaptive institutional and economic solutions for agricultural management to reduce the risk of aquifer contamination to set up.

#### The Korba-El Mida artificial recharge site

The artificial recharge system consists of three rectangular pools ( $50 \times 30 \text{ m}$ ) dragged to a depth of 1.5 m to be recharged with a daily volume of  $1500 \text{ m}^3$  with the wastewater treatment plant (WWTP) discharge of Korba. The recharge site of Korba-El Mida is located at 300 m in the north of the WWTP and at 1.5 km far from the coastline (Fig. 1). The WWTP collects domestic and industrial water including primary, secondary, and tertiary treatments. The treated wastewater (TWW) is recharged according to a gravity flow from a  $300 \text{ m}^3$  nearby storage reservoir. The bottom of the pools is covered by a 0.5-m layer of sand in order to retain the suspended materials. The recharge rate is quantified based on the volume of TWW discharged in each pool and the corresponding number of hours followed on site by the manager. Infiltration rates were measured in field using double ring infiltrometer of 30 and 55 cm in diameters. The rate was calculated according to the measurement of water levels in time until rate stabilization. However, this infiltration rate is lower due to evaporation and clogging in the vadose zone. Jarraya Horriche and Benabdallah (2020) estimated that infiltration should be reduced by 10% of the discharged volume in the pool.

The artificial recharge started in December 2008, and the recharge facility was initially designed for a daily and continuous filling of the three pools with TWW in an alternate scheme. However, due to the suspended materials and the organic loads in the TWW, the filtration system is clogged and the operational system is turned out to a manual functioning.

The water table nearby the pools is at about 15-m depth from the soil surface. The unsaturated zone is



**Fig. 1** Location of the study area. Location of Tunisia (a) and the Cap Bon (b). Location of the treated wastewater artificial recharge site (TWWARS) and the freshwater artificial recharge (AR) and geological setting (c)

formed mainly by sand with thin layers of sandstone with high infiltration rate ranging between 1 and 65 m/day (CRDAN 2004).

#### Approach for setting groundwater monitoring network

Prior to the TWW artificial recharge activity, we conducted an investigation covering 180 wells during February to March 2009 (164 shallow wells, 15 piezometers, and 1 deep well) (Fig. 3). The collected information was set up in a database (Fig. 4a–d) which is related to the well’s geographic location, characteristics, equipment, exploitation, groundwater level, and water

quality. Among these wells, a monitoring network was selected all around the recharge site for a radius of about 2 km (Fig. 3). We used several criteria in order to select the best representative monitoring network of groundwater levels and quality. In fact, their variations are dependent on multiple natural and management factors such as the rainfall, evapotranspiration, groundwater pumping, and artificial recharge. They are also dependent on the soil, land use, vadose zone, and hydrogeologic settings which define the infiltration and groundwater flow.

The water table depth (WD) ranges between 2.5 and 30.9 m (Table 1), and water salinity (S) ranges between

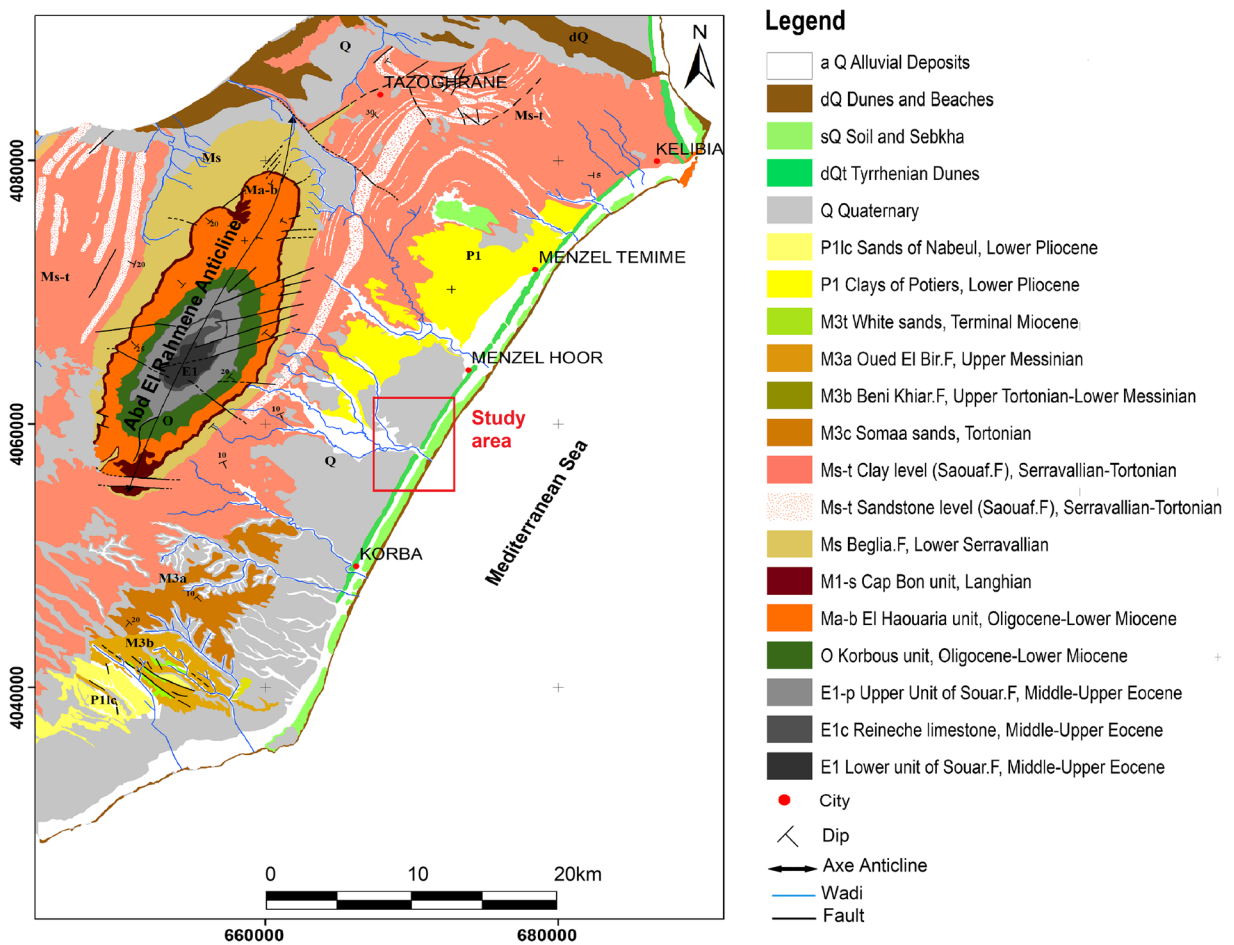


Fig. 2 Geological map of the Dakhla syncline

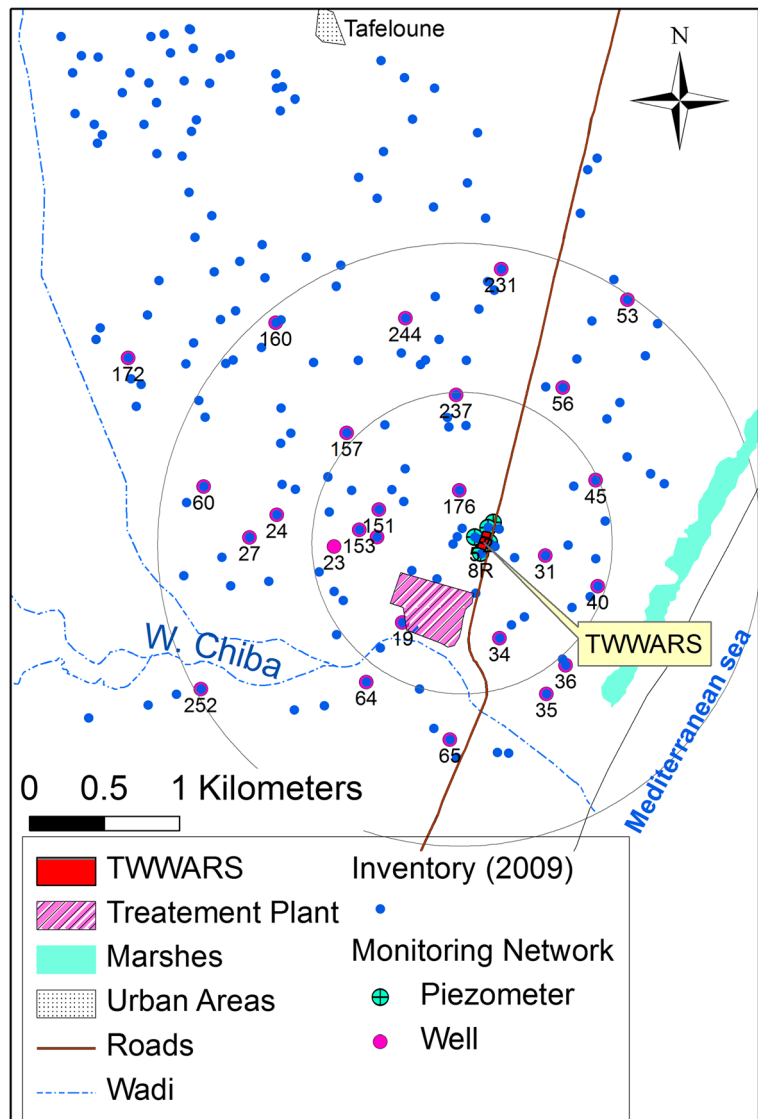
0.3 and 12 g/L. Water level (WL) ranges between -9.2 and 6.1 m. Near the artificial recharge site, the WD is about 15 m/soil. The lowest WL is recorded in the inland zones upstream the artificial recharge site corresponding to the irrigated areas. WL is above sea level near the coastline which matches with the sandy dunes favorable to rainfall recharge. A particular attention has been paid to the quality of the data while considering the setting of the monitoring network. The data were checked against errors that can rise from the use of inappropriate measurement devices (e.g., non-professional GPS), reference elevations, wells locations, time and pumping rates, etc.

A clear axis of high salinity values exceeding 4 g/L was also observed near the overexploited areas in the direction of the sea. However, low groundwater salinity was registered in some regions near the coastline due to freshwater recharge in the sandy dunes. Water salinity

was about 4 g/L near the site before the start of recharge. It is worth noting that this value was taken in the month of February corresponding to a wet period.

Thus, we first considered the groundwater flow and water salinity gradients in the location of the recharge site when setting the monitoring network. A series of wells were selected between the site and those indicating low or high water levels and salinities. A radius of 2 km was chosen all around the site to cover the influenced zone of the artificial recharge. The density of the selected wells is higher for the first 1-km radius. One well was located over the 2-km radius but was included as a control well in order to assess the non-influenced area. Others were selected along the right side of Chiba wadi in order to assess the possible impact from the river recharge in this area. Some wells, located in the sandy dune areas, were also selected as they are favorable for rainfall recharge. Piezometers close to the site were also

**Fig. 3** Location of inventory wells and monitoring network for the TWWARS

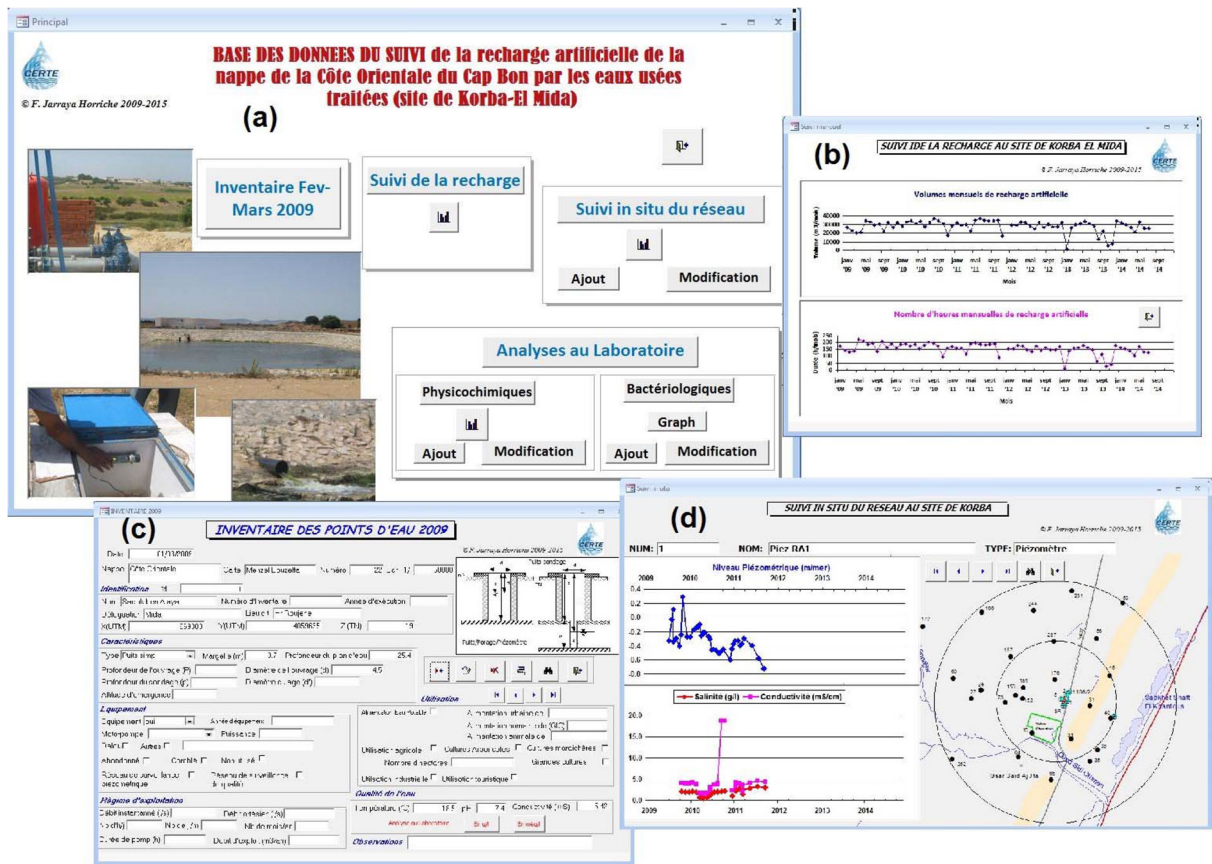


selected in order to study the immediate impact. The monitoring network was finally chosen by considering the wells' locations, utilization, and accessibility. Some wells were replaced by others during the monitoring when technical problems occurred. The new wells are close to the replaced ones with similar water levels and salinities. In fact, well 19 was removed from the monitoring network as we discovered that the farmer deepened this well to catch the deep aquifer. The final monitoring network is made of 25 traditional wells and 5 piezometers (Fig. 3) and sampled bi-monthly since June 2009 for water level depth, water salinity, conductivity, temperature, and pH in situ measurements. Water salinity is deduced from in situ hydraulic electrical

conductivity and temperature measured by WTW probe during the 48 campaigns from June 2009 until September 2012. Measures started according to bi-monthly frequency but decreased from March 2012.

Since September 2011, the two piezometers PZ1 and PZ5 were equipped with data logger and multi-parameter sensors for measuring groundwater level, salinity, conductivity, dissolved oxygen, pH, and redox potential. This continuous flow of information is used to assess the immediate impact of the artificial recharge.

Land elevation of the monitoring network was measured using professional GPS to reduce errors in the WL calculation.



**Fig. 4** Data base for TWWARS monitoring. (a) Main menu. (b) Graphic for historical recharge volume and duration, (c) Introduction of well inventory data. (d) Interactive water level and salinity investigation for the monitoring network

**Results and discussions**

The artificial recharge

During the time period between January 2009 and November 2014, the daily recharge volume ranged between 0 and 1745 m<sup>3</sup>, while the recharge duration ranged between 0 and 10 h a day. The cumulative water

volume during this time period reached 1.8 MCM for 1335 days of operation (Table 2). No recharge days represented 38% for the investigated time period leading to a daily recharge time duration of 8 h for about 37% of the days. The overall variation in recharge rates and duration are synthetized by the boxplots presented in Fig. 6 a and b. The latter shows the decline in the rate of recharge and the number of days for TWW discharged

**Table 1** Statistical data of the well inventory (February to March 2009)

Parameter	Water depth (WD) in m/swl	Water level (WL) in m/swl	Salinity (S) in g/L
Medium	19.6	-1.1	3.1
Minimum	2.5	-9.2	0.3
Maximum	30.9	6.1	12.0
Standard deviation	5.8	2.7	1.7
Variance	33.4	7.2	3.0
N	167	162	132
Confidence level (95.0%)	0.9	0.4	0.3

to the pools. The wide spread of the different boxes are mainly due to clogging problems as confirmed by Vandenbohede et al. (2009).

The recharge rate of TWW in the pools ranges between 138.8 and 202.3 m<sup>3</sup>/h (Table 2). It ranged between 0 and 36,840 m<sup>3</sup>/month (Table 2 and Fig. 5) which is lower than the designed volume of 45,000 m<sup>3</sup>/month (CRDAN 2004). This is mainly due to the changes in the recharge duration.

In order to check the efficiency of the recharge, the effective infiltration rate was measured in April 2013 and June 2013, and it was equal to 2.9 and 1.3 m/day, respectively. These values are lower than the calculated values in 2004 which ranged between 1 and 65 m/day (CRDAN 2004). This can be due to a clogging layer formed on the pool's surfaces after 3 years of recharge. In fact, clogging was also observed after artificial recharge in two cases in Germany and in Belgium (Greskowiak et al. 2005; Vandenbohede et al. 2009). The authors attributed the clogging to the change in the water quality that resulted from the interaction of physical filtration, chemical sorption or precipitation, and biological processes in the pond and to the algae accumulation on the bottom of the infiltration surface.

The TWW salinity, obviously better than that of the groundwater, is intended to create a barrier against the seawater intrusion and improve the groundwater quality upstream in the agricultural areas. Many authors investigated the TWW salinity. It ranged between 2 and 3 g/L in 2008, reaching exceptionally 6 g/L in August 2008 (El Ayni et al. 2011). During the time period of 2008–2012, salinity ranged between 2 and 5 g/L (Gaaloul et al. 2012; Ouelhazi et al. 2014). However, groundwater salinity was equal to 10 g/L in the location of the artificial recharge site in August 2008 with high chloride concentration (El Ayni et al. 2011).

#### Water level monitoring

The WL and salinity measurement frequency was set to bi-monthly from June 2009 to March 2011, and then it was set to a monthly frequency due to budget availability. The network was not monitored during the period between February 2013 and April 2014. Overall, 50 monitoring campaigns were realized, and the number of monitored wells for each campaign ranged between 18 and 27 in the majority of the situations. About a thousand of measurements for groundwater level were performed and saved in a database. The collected

groundwater level and salinity were analyzed considering temporal and spatial variations. Several anomalies were first detected and corrected or removed. Such anomalies occurred when a pumping is in progress during the WL measurement or because of a change in the original of measurement. The validated WL was then saved to the database presented in Fig. 4 and then used for interpretation.

During the monitored period, we noticed the different behaviors of groundwater level variations. Along the coastal fringe up to the location of the recharge site, for less than 2 km, WL varies according to the seasonal fluctuations not more than  $\pm 0.5$  m and with a quasi-stability or a slight decrease. WL variations were due mainly to climatic conditions. PZ1 located close to the site was surveyed bi-monthly until September 2011 and continuously afterward. Figure 7a presents the observed WL together with the rainfall patterns. It seems that the artificial recharge did not have a detectable influence on WL even during dry periods. This result is in contrast to other findings in different study areas (Greskowiak et al. 2005). While a slight increase of WL is registered upstream the recharge site as shown by wells 24 and 27 in Fig. 7b, an increase in water salinity led to the abundance of many wells within this area. In the right side of the Chiba wadi, WL is almost stable and below seawater level. It varied according to the seasonal fluctuations with about 1 m of amplitude as demonstrated by well 252 (Fig. 7c). The mapping of WL during wet and dry seasons (Figs. 8 and 9) confirms that this region corresponds to the overexploited area where WL decreases to  $-6$  m. Compared with the initial situation in 2009, the variation maps show an increase in WL from the artificial recharge site during wet periods (Fig. 10a). This increase is due to the decline of pumping as many farmers are no longer cultivating their lands. However, a generalized decrease in WL of about 1 m is registered elsewhere, confirming that artificial recharge did not affect significantly the groundwater level.

#### Groundwater salinity monitoring

The results revealed a spatial pattern in salinity distribution. The salinity was very high near the coast as revealed in wells 35, 36, and 40, which could be linked to the saline water intrusion. These high salinity values were exacerbated mainly when the water is pumped following the upconing of seawater as shown in well 31 (Fig. 7d). On the other hand, the salinity is lower than



**Table 2** Statistical data of TWW artificial recharge (January 2009 to November 2014)

Parameter	Recharge flux (RF)			Recharge duration (RD)		
	Unit	$m^3/d$	$m^3/h$	$m^3/month$	$h/d$	$d/month$
Medium		851.1	181.3	25 881.3	4.7	18.8
Minimum		0.0	138.8	0.0	0.0	0.0
Maximum		1 745.0	202.3	36 839.0	10.0	27.0
Standard deviation		695.5	13.3	9 951.0	3.8	7.2
Variance		483 662.3	176.8	99 022 814.5	14.5	52.2
N		2 159	1 335	71	2 158	71

2 g/L in the sandy dunes and about 4 g/L in the PZ5 located close to the recharge site. It is worthy to remind that the sandy geologic features in the former location are a permeable zone with high potential for freshwater recharge. By going further from the sandy dunes and pointing to the north of the recharge site, the water salinity increases and ranges between 6 and 14 g/L (Fig. 7 e and f). However, by going westward from the recharge site, the water salinity ranges between 5 and 6 g/L near wells 151, 152, and 153 and between 3 and 4 g/L near wells 24 and 27. In fact, this area corresponds to the overexploited zone where groundwater flows toward the piezometric depressions. The maximum salinity decrease is observed in well 176 reaching 10 g/L during all the monitored period.

The spatial mapping of water salinity during the wet and dry seasons is presented in Figs. 11 and 12. The maps confirm that higher salinity values are recorded in the northern part of the recharge site with the highest values recorded at the sampled

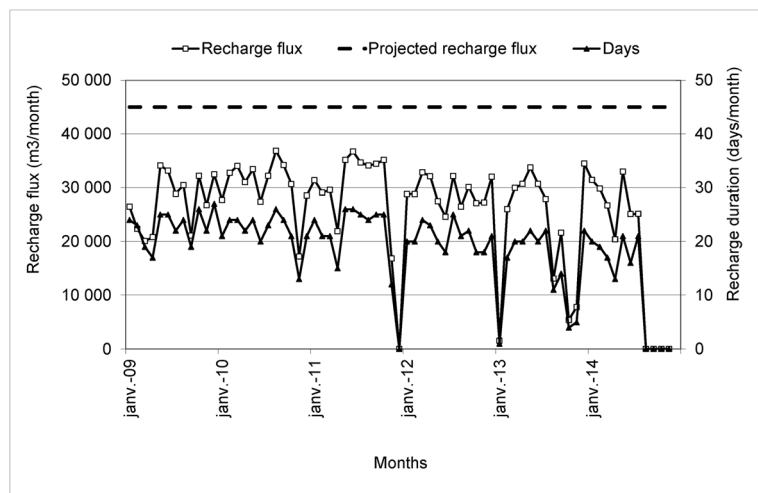
well 176. Water salinity exceeds 4 g/L in a large area and is more pronounced in the dry season than in wet season. This could be related to the increased pumping activities during the dry season combined with the saline water intrusion. Conversely, lower salinity values are measured in wells located at the sandy area along the coastline. The decrease in salinity in these wells is likely to be related to the decrease in the pumping activities.

The decrease in seasonal water salinity is spreading over a radius of 5.5 km<sup>2</sup> from the recharge site (Fig. 13). The highest decrease is recorded at well 176 where the salinity decreased from 14 g/L in February 2009 to 4 g/L in September 2012.

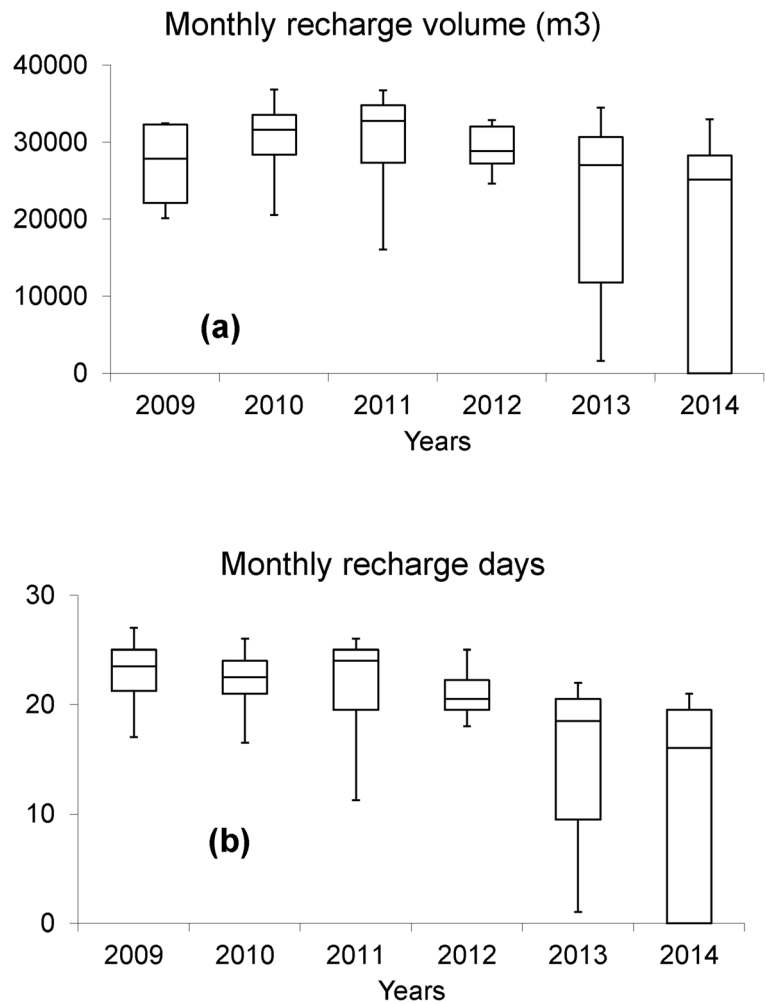
Discussion

Although the recharge activity has been performed in 6 years, it seems ineffective in improving water level in the Korba-El Mida site. This result is due

**Fig. 5** Variation of the TWW artificial recharge flux and duration

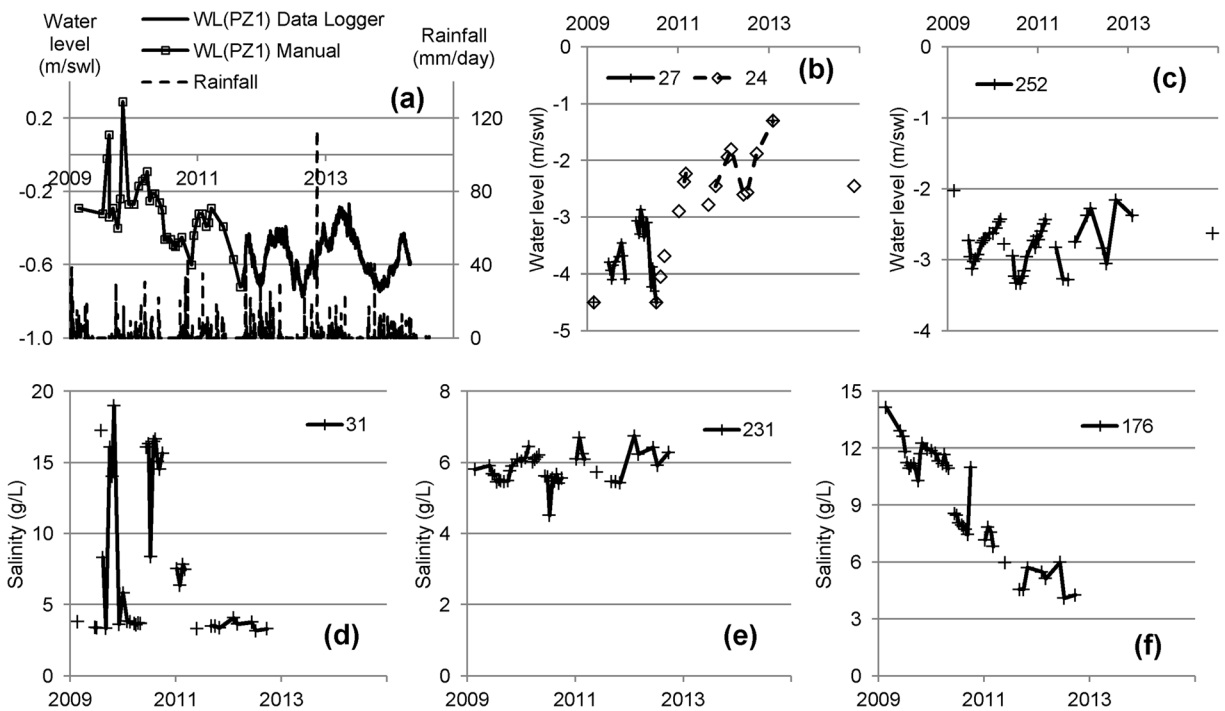


**Fig. 6** Monthly recharge rate (a) and duration (b) analysis



to the location of the site close to the sea which is a Dirichlet boundary condition for the groundwater flow. According to the hydrogeologic study performed in 2004 (CRDAN 2004), the groundwater flow was equal to 0.6 m/day near the recharge site. If we consider the recharged volume between February 2009 and February 2013, the average recharge rate remained about the same. The infiltration rate is normally lower than this value because of the frequent clogging periods, losses in the vadose zone, and evaporation. It is more likely that the performed artificial recharge rate was unable to have positive impacts on groundwater flow and could not improve groundwater level raise (Fig. 10b). In addition, there is no clear evidence that the small increase registered in the upstream area of the site is related to the recharge activities.

It seems more linked to the decline in the pumping activities due to the bad water quality rather than to the artificial recharge. In fact, this was confirmed during the field surveys. The local farmers pump water that was transferred from other locations through the nearby channel. While Ouelhazi et al. (2014) claimed that the increase in WL recorded at well 60 is a direct impact of the recharge activities, the current study reveals that this increase is related to the seasonal variation rather than a positive effect of the artificial recharge. For instance, WL as monitored in the aforementioned well showed a decrease in 2013 and 2014, which clearly confirms the ineffectiveness of the artificial recharge activity due to the low actual recharged volumes reaching the aquifer



**Fig. 7** Variation of groundwater levels (a, b, c) and salinities (d, e, f). Variation of rainfall is added with groundwater level for PZ1 (a)

and the reduced infiltration rate following the phenomena of evaporation and losses in vadose zone.

In addition, results from wells 27 and 24 showed that the increase of 3.2 m in WL in 2009 and 2013 was very local and, therefore, could not be attributed to the artificial recharge activity. It is more a consequence of the reduction in the water pumping activity. This statement was also reported by the local farmers.

In a recent modelling effort by Jarraya Horriche and Benabdallah (2020), a 2D flow and solute transport models were used to evaluate the impact of this constructed artificial recharge ponds on groundwater level and water salinity and to predict its impact based on several management scenarios. Several sensitivity tests for the effective artificial recharge were then performed for the period between 2009 and 2014. Thus, it was shown that the best simulation that reproduced the water level and the water salinity closest to the artificial recharge site in piezometer P3 was based on considering that only 70% of the discharged flow into the basin is able to gain the aquifer, with an infiltration rate decreasing linearly between 2009 and 2014 with a negative slope of 0.12% and an input of 1 g/L for TWW salinity. The decrease in

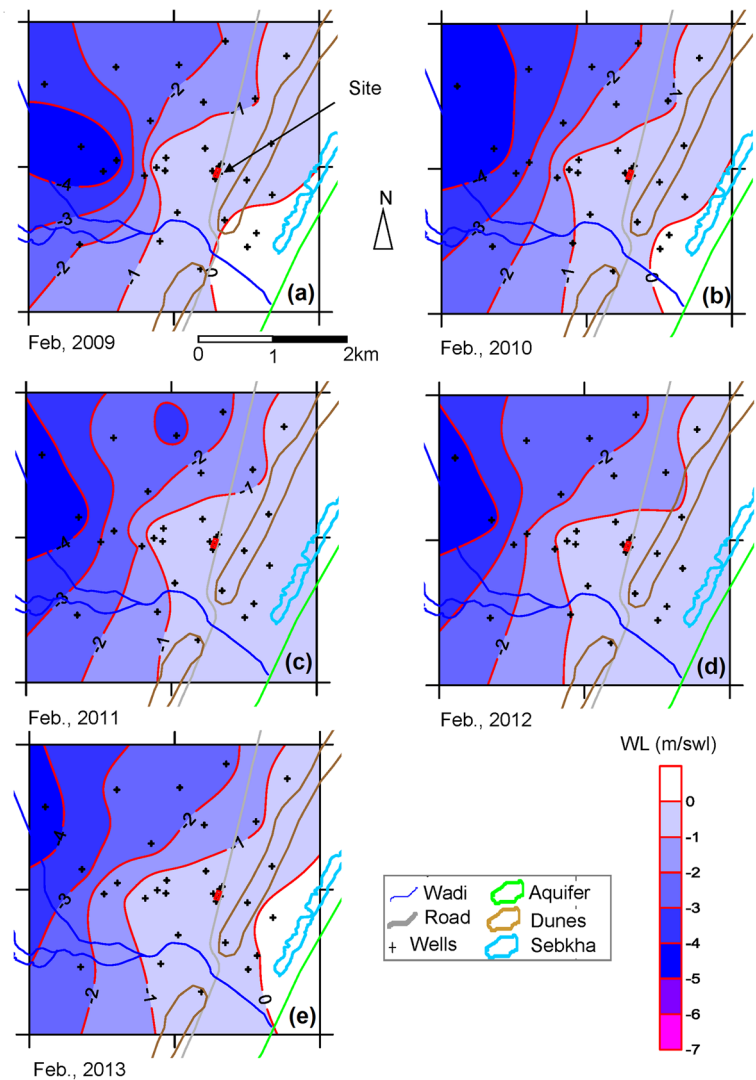
infiltration rate was used to take into account the partial clogging of the soil and the vadose zone. The decrease in TWW salinity is considered, following the purification of TWW by the unsaturated zone.

Figure 14 reports the simulated groundwater level by the end of 2009 near the study site and salinity variation between 2009 and 2012. We note the high similarity with the observed map (Fig. 8a). It also confirms that groundwater flows from the sea toward the inland areas and the wadi did not contribute to the drainage of groundwater since groundwater level is far less than the elevation of the wadi's bottom.

Under other conditions, Valhondo et al. (2020) presented six artificial recharge pilot systems to study the role of diverse TWW parameters for artificial recharge. They noted that the installation of reactive barriers in the recharge areas favors the performance of the systems and the combination of both plants and reactive barrier in the recharge zone seems to favor the infiltration rates by preventing clogging. Such options could enhance the infiltration rates and its influence on groundwater recharge.

On the other hand, the artificial recharge seems to have an effect on the groundwater salinity as

**Fig. 8** Groundwater level maps during wet seasons (2009–2013)



shown by the monitoring results. In fact, the physical processes controlling the solute transport in porous media are due to both advection and hydrodynamic dispersion (Freeze and Cherry 1979). Even though the recharge did not affect the groundwater flow, it was effective in reducing water salinity. However, the impact of the recharge had lower influence during the wet seasons compared with precipitation. Furthermore, the salinity in wells 24 and 27, located upward and outside the 1-km radius, is stable despite the state of groundwater level, confirming that they are not affected by the artificial recharge. However, an increase in salinity is observed in wells 23, 153, and 244 located inside the 1-km radius. Near the

coastal line, the wells present conflicting salinity trends due to the impact of upcoming phenomena as presented in Bear et al. (1999).

Comparing the monitoring results with the simulated groundwater and solute transport models (Jarraya Horriche and Benabdallah 2020), we note that salinity registered a trend of decrease, between 2009 and 2012 for the surrounding area of the artificial recharge site (Fig. 14b), which is comparable to the monitored variation in Fig. 13b but with a lesser magnitude. This result endorses in part the fact that the decrease in groundwater salinity is related to the artificial recharge, but it could also be related to other factors such as the stopping of groundwater pumping by farmers.

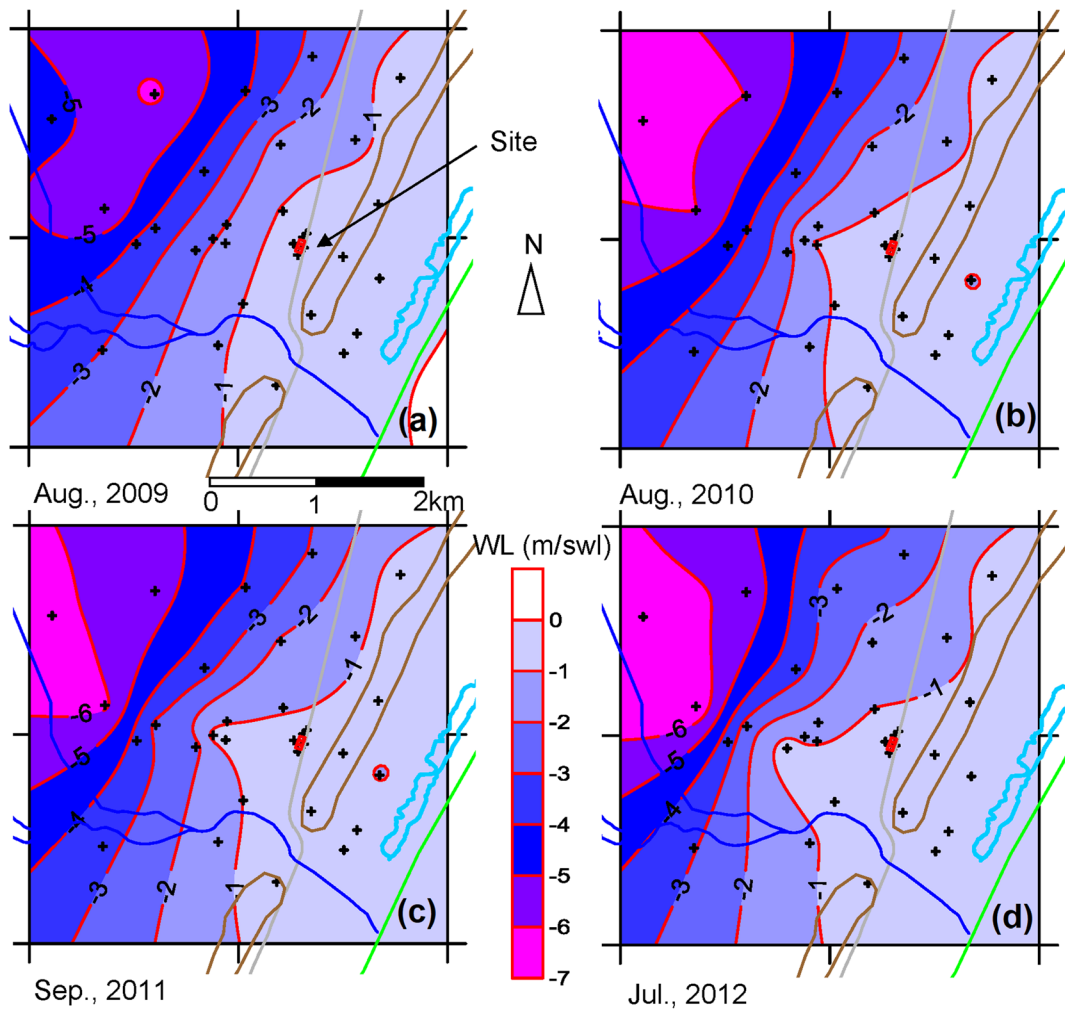


Fig. 9 Groundwater level maps during dry seasons (2009–2012)

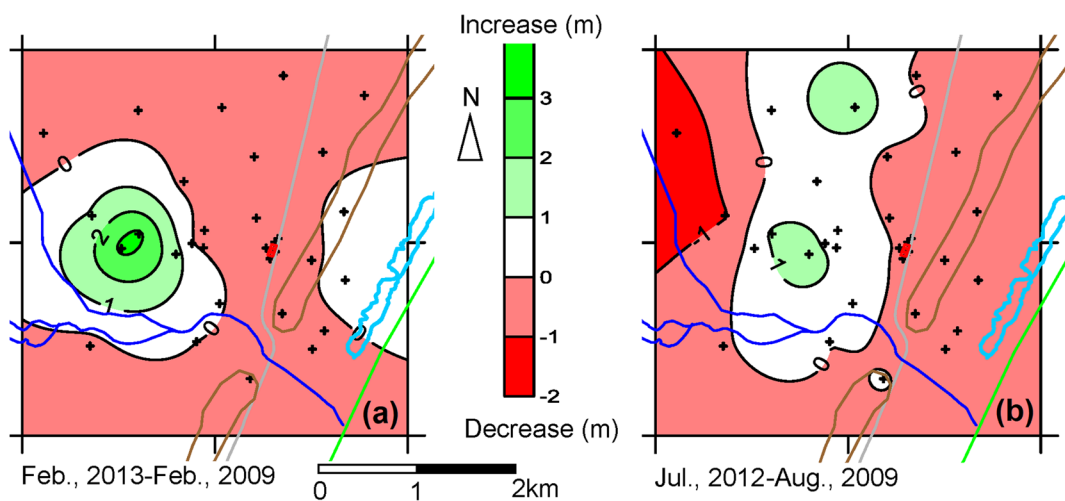
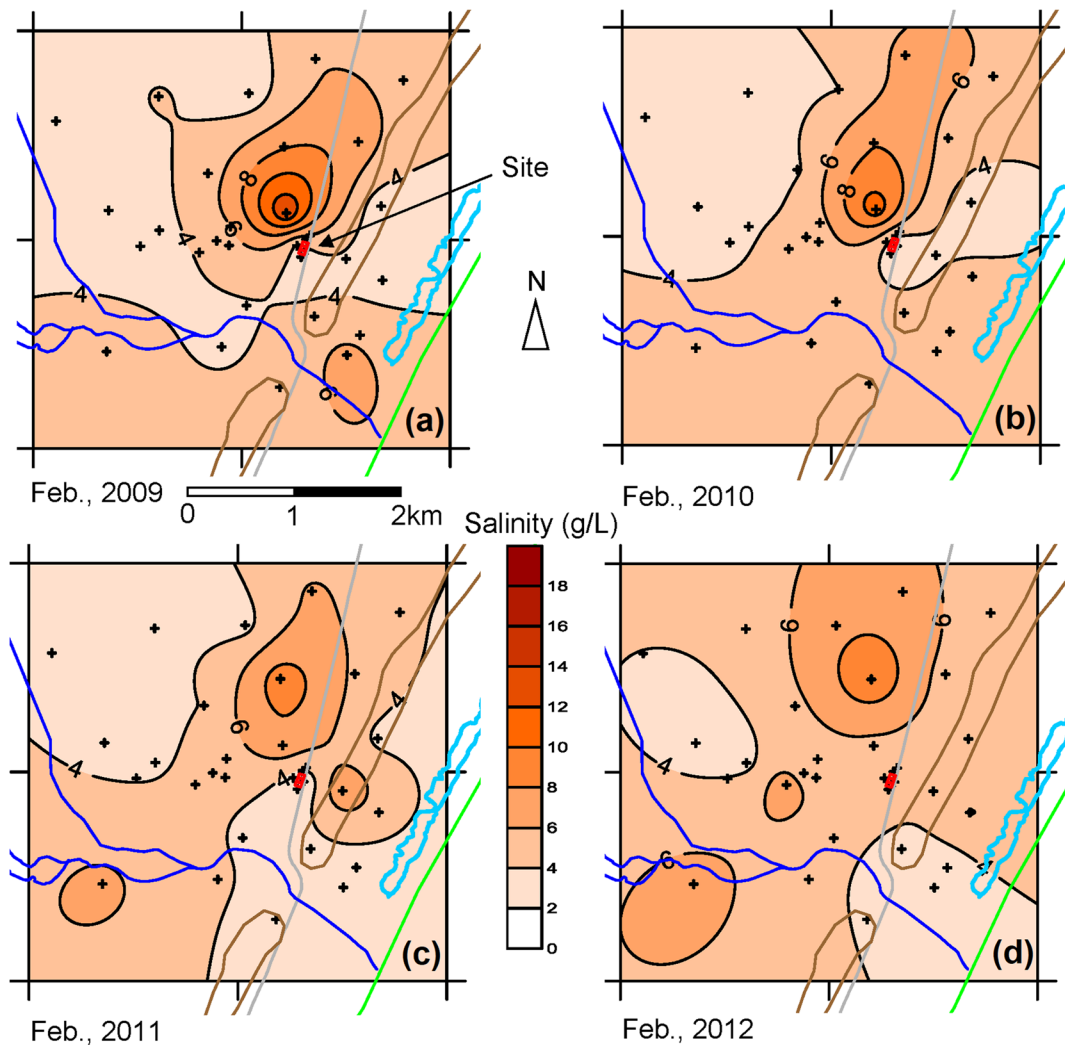


Fig. 10 Groundwater level variation maps during wet seasons (a) and dry seasons (b)



**Fig. 11** Salinity maps during wet seasons (2009–2012)

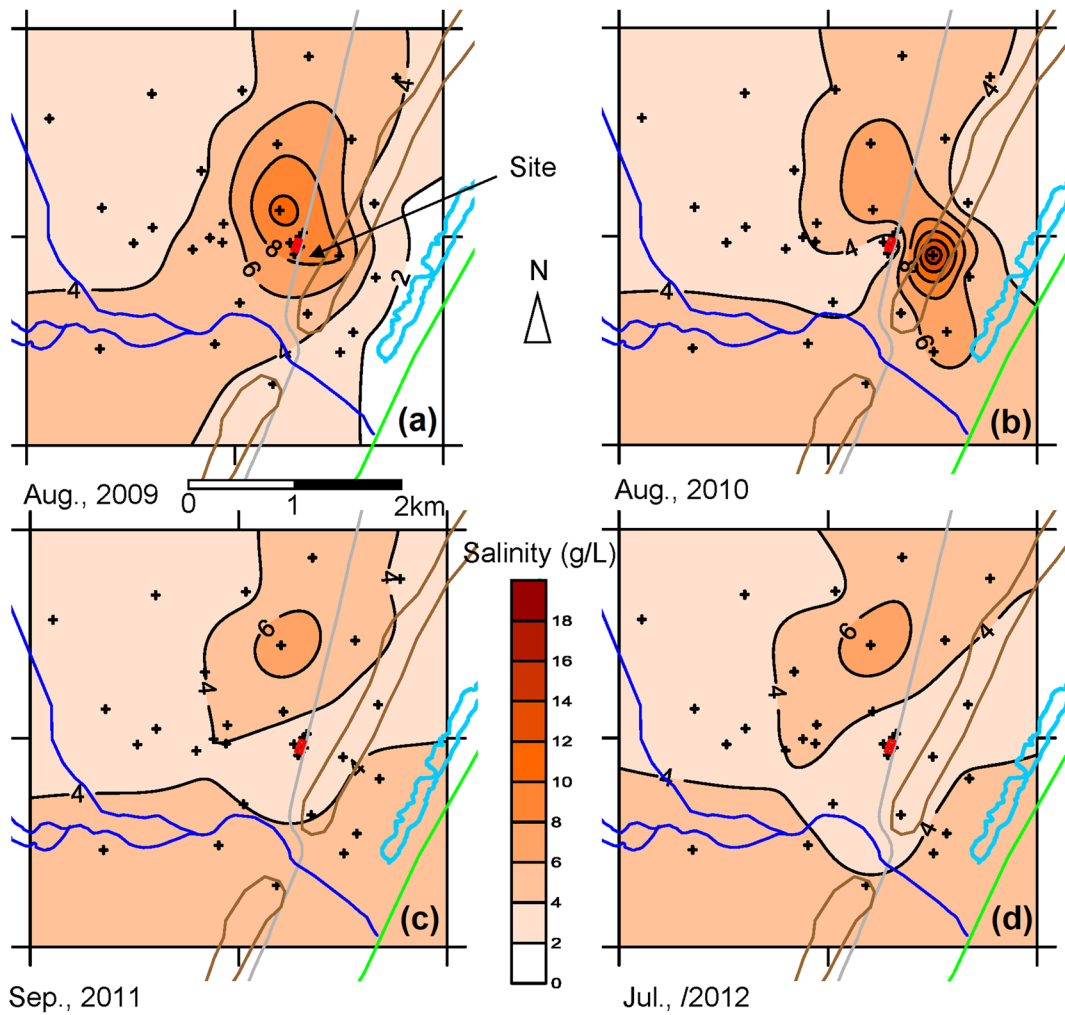
## Conclusions

Even though there are diverse direct and indirect approaches and advanced models for groundwater recharge estimation presented in an overview of the different techniques by Singh et al. (2019), direct measurements of water table and water quality fluctuations are still a valuable mean for understanding short- to long-term recharge alterations.

In the Korba aquifer, groundwater artificial recharge using TWW was applied to overcome the salinization of this coastal aquifer. However, the analyses of the 6-year monitoring reported several issues with the recharge system including clogging of the filtration medium, reduction in the recharge rates as compared with the

one initially planned, etc. Thus, the implementation of recharge pools in a semi-arid area with low recharge rates showed its limitations in improving the WL. Despite the fact that a reduction in water salinity was recorded in many investigated wells, the site investigation showed that this decrease is more related to the decrease in the groundwater pumping frequency as many farmers abandoned their wells. This was also confirmed by the low decrease of the salinity both in time and space.

This study stresses the urgent need for reconsidering the operational guidelines of the recharge system by planning periodic and systematic cleaning of the pools and spreading a layer of fine sand in order to reduce the clogging phenomena as well as to seek a further improvement for the TWW quality. A recent study by

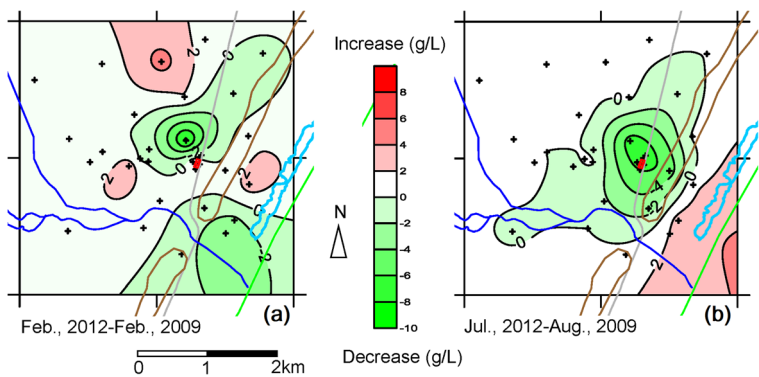


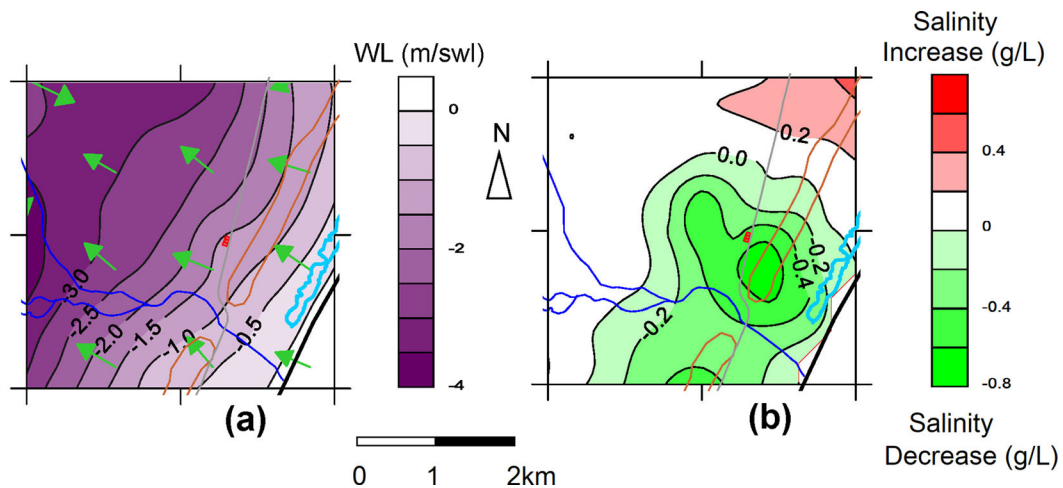
**Fig. 12** Salinity maps during dry seasons (2009–2012)

Valhondo et al. (2020) showed that the presence of plants in the pools seems to favor infiltration by preventing clogging. Before replicating this technique on other Tunisian coastal aquifers as is projected, further

work is needed to clarify a number of issues related to the used technique, the frequency and the period of the monitoring network, and the coupling with other techniques of recharge estimation.

**Fig. 13** Salinity variation maps during wet seasons (a) and dry seasons (b)





**Fig. 14** Validation of flow direction (a) and salinity variation (b) by groundwater modelling

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