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Article

Evaluation of the Groundwater Quality Using the Water Quality Index and Geostatistical Analysis in the Dier al-Balah Governorate, Gaza Strip, Palestine

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Abstract: Groundwater contamination is a major problem in the Gaza Strip. In this study we investigate the groundwater quality in the Dier al-Balah Governorate. Water samples were collected from 19 municipal wells in April 2009 and April 2014 and analyzed for physio-chemical parameters (pH, TDS, Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, Cl^{-}, SO_4^{2-}, HCO_3^{-} and NO_3^{-}). The aim of the research is to determine the groundwater quality and to produce groundwater quality maps using the water quality index (WQI) method and geostatistical analysis. The results show that all water samples are very saline due to the intrusion of Mediterranean seawater in the coastal aquifer. Differences in chemical composition between 2009 and 2014 indicate that about 1% more seawater was mixed with the groundwater in this period. The majority of the observed chemical parameters of all wells are well above the WHO water quality standards and all WQI values indicate that the water quality is problematic. The spatial variation of the WQI scores is modelled by a deterministic component expressing a linear dependence on the distance to the coastline and a stochastic residual described by an exponential variogram with a practical range of 3000 m. The mapping of the WQI scores and derived water quality classes is achieved through regression-kriging. The results indicate that the groundwater in a large area along the coastline is unsuitable for human consumption and comparison of the maps of 2009 and 2014 shows that this region further expanded by about 700 m inland in a period of 5 years. The results of this study are worrying, but they also contribute to a better understanding of the factors that determine the groundwater quality and can help authorities and stakeholders with sustainable development.

Keywords: groundwater quality; water quality index (WQI); regression-kriging; seawater intrusion; Gaza coastal aquifer; Palestine

1. Introduction

Groundwater is an important source of fresh water for human consumption, irrigation and industrial use in many countries of the world. However, residential, industrial, commercial, agricultural and other anthropogenic activities together with natural conditions often lead to a deterioration in groundwater quality [1,2]. That is why an assessment of the quality of groundwater is of great importance for society. Water quality assessment includes an evaluation of the physical, biological and chemical properties of water in relation to the natural quality, intended use and human effects that can influence the health of aquatic systems [3].

More than 90% of the population in the Gaza Strip is dependent on the municipal drinking water network, while the remaining 10% of the population rural areas is dependent on private sources [4]. The quality of groundwater in the Gaza Strip is influenced by many processes, including return flow from irrigation, wastewater leakage, upconing of underground brine water and sea water intrusion [5,6]. Shomar [4] found that only 10% of the municipal wells meet WHO drinking water standards; in particular the chloride, nitrate and fluoride concentrations are 2–9 times higher than WHO standards. Almasri and Ghabayen [7] reported that a large portion of nitrate concentrations observed in wells in the Gaza Strip did not meet the US Environmental Protection Agency drinking water standard for the years 1990 and 2000–2004. Chemical analysis of municipal wells conducted in 2009 by Aish [8] showed that total dissolve solids, chloride and sodium values exceeded WHO standards. Abbas et al. [9] sampled 58 municipal wells in 2010 and found that no groundwater in the Gaza Strip meets all WHO drinking water standards. Alastal et al. [10] derived groundwater quality maps for the Gaza Strip using six water quality parameters (chloride, nitrate, sulfate, calcium, magnesium, and alkalinity) and found that the fraction for which the groundwater quality is not good increased from 30% to 55% between 2000 and 2010. The Palestinian Water Authority [11] reported that only 12.4% of the wells comply with the WHO standard for nitrate and only 19.3% with the WHO standard for chloride. Abu-alnaeem et al. [12] reported that around 90% of all wells in the Gaza Strip are polluted by nitrate, mainly due to wastewater inputs.

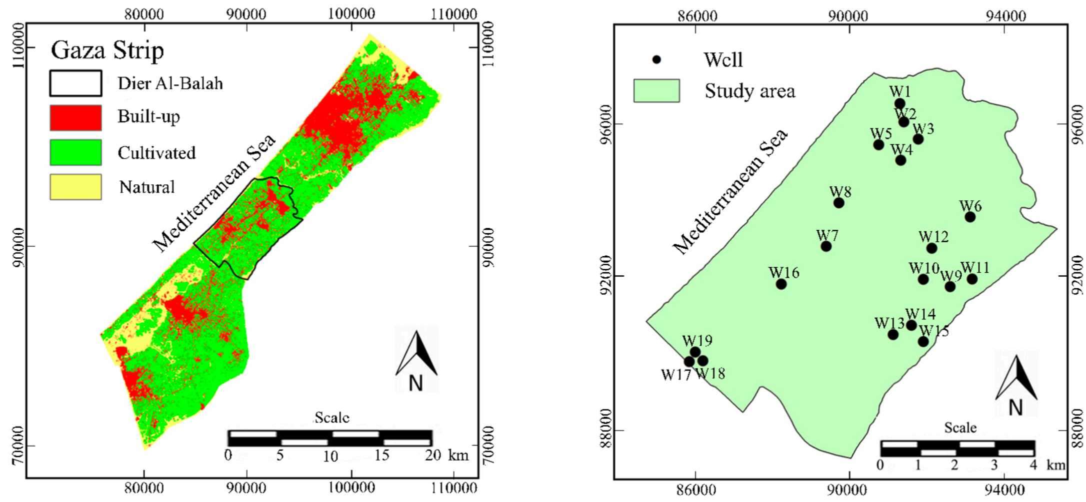
To assess the suitability of groundwater for human consumption, it is essential to determine and evaluate its quality. Researchers have used different methods to express water resources quality. Traditionally, water engineering professionals compare individual chemical parameters with recommended allowable limits. In many regions with scarce water resource, however, the use of water at concentrations slightly above these limits is generally not harmful. Horton [13] proposed a water quality index (WQI) to describe the suitability water for human consumption in a single score that can be ranked into categories using terms such as excellent, good, poor, very poor and unsuitable for use, which are easy to understand for decision makers and consumers. Various methods have been proposed to derive WQI scores [14]. A weighted WQI score is usually used in which ratios of concentrations of water quality parameters and their recommended standard values are weighted and combined in a single number. Recent applications of the weighted WQI approach in groundwater quality studies have been presented [10,15–18]. All methods used to derive WQI scores are similar, the only difference being the number of parameters (observations) used and their corresponding weights.

The purpose of this study was: (1) to assess the groundwater quality in the Dier al-Balah Governorate of the Gaza Strip using groundwater samples taken from municipal wells in 2009 and 2014, (2) to evaluate the suitability of the groundwater for human consumption using the WQI method, (3) map the spatial distribution of the groundwater quality, and (4) evaluating the evolution of the groundwater quality for the 2009–2014 period.

2. Materials and Methods

2.1. Study Area

The Dier al-Balah Governorate is located in the center of the Gaza Strip (Figure 1a). The governorate is about 10 km long and 5.5 km wide, with a total land area of 55.2 km^2 and covers 15% of the Gaza Strip. Located in a coastal zone, transitional between the temperate Mediterranean climate and arid climate of the Negev and Sinai deserts, the Gaza Strip experiences a semi-arid climate with two well-defined seasons: a wet season from October to April and a dry season from May to September. The average daily temperature ranges from 27 °C in summer to 13 °C in winter and the average annual rainfall is about 335 mm/y [19]. In 2017 around 273,200 people lived in the Dier al-Balah Governorate resulting in a population density of 4949 persons per km^2. In recent decades, the demand for water for domestic purposes has increased noticeably as a result of the high population growth.



(**a**) (**b**)

Figure 1. Map of the Gaza Strip with indication of the Dier al-Balah Governorate (**a**); map of the Dier al-Balah Governorate (study area) with the location of the municipal wells used in this study (**b**).

The Dier al-Balah Governorate is generally flat with an average topographic elevation of about 45 m above the mean sea level. The soil type is mainly sandy loess or loessal sand in addition to some sandy regosols along the coast [20]. Land use is dominated by agricultural land, urban and built-up areas, natural reserves and recreational areas in the coastal part [20]. The coastal aquifer in the Dier al-Balah Governorate belongs to Pliocene-Pleistocene age and consists of Kurkar Group deposits of calcareous sandstone, silt, clay, unconsolidated sands and conglomerates [21]. The aquifer is the only natural water resource and is used for various purposes, such as drinking water, crop irrigation and industrial uses [10].

2.2. Data Collection and Procedures

In April 2009 and April 2014, water samples were collected from 19 municipal wells in the Dier al-Balah Governorate (Figure 1b) by the first author on behalf of the Coastal Municipalities Water Utility, responsible for the production of drinking water. The depth of the wells varies from 60 m to 90 m. Groundwater quality parameters were analyzed using standard procedures and suggested precautions were taken to prevent contamination [22]. During the groundwater sampling, temperature, pH and electrical conductivity (EC) were measured in situ according to WHO guidelines [3,23]. Total dissolved solids (TDS) was derived from EC using a calibrated linear relationship. Concentrations of major cations (Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}) and anions (HCO_3^{-}, SO_4^{2-}, Cl^{-} and NO_3^{-}) were determined in the laboratory of the Coastal Municipalities Water Utility according to standard methods prescribed by the American Public Health Association [22]. For quality assurance and quality control, duplicate tests were performed throughout the laboratory analyses.

An indication of the accuracy of the water analysis can be obtained using the charge-balance equation based on the electroneutrality condition. The deviation from equality is expressed as:

$$E = \frac{\sum \text{cations} - \sum \text{anions}}{\sum \text{cations} + \sum \text{anions}} \times 100\% \qquad (1)$$

where E is the charge-balance error, and $\sum$ cations and $\sum$ anions are the sums of cations and anions, respectively, expressed in equivalents per liter. The calculated charge-balance error must be within an acceptable limit of ±10% [24]. The major cation and anion compositions of the groundwater samples are presented in a piper diagram [25], with which hydrogeochemical facies can be categorized and trends in the data can be detected.

2.3. Water Quality Index

The derivation of WQI involves weight assignment for each parameter, normalization of the weights, standardization of the parameters, calculation of individual WQI scores and aggregation of the scores. Weights for the parameters (pH, TDS, Ca^{2+}, Mg^{2+}, Na^+, K^+, HCO_3^-, SO_4^{2-}, Cl^- and NO_3^-) given in Table 1 were adapted from previous studies [10,16]. The weights values range from one to five based on their importance for water quality assessment. Total dissolved solids and nitrate concentration are given the maximum weight of five because these are important indicators for assessment of the overall water quality, while potassium is given the minimum weight of one because it poses little risk to consumer health. Other chemical parameters are assigned a weight between one and five depending on their importance in determining the water quality. A factor of three is assigned to pH because of the indication of alkalinity and effect on hardness and to sodium, chloride and sulfate because of their indication of salinity and effect on taste. Calcium, magnesium and bicarbonate are given a weight of two because of their effect on hardness.

Table 1. List of physico-chemical parameters with corresponding unit, WHO standard (S_i), WQI weight (W_i), and normalized weight (w_i).

Chemical Parameter	Unit	Standard	Weight	Relative Weight
		S_i	W_i	w_i
pH	-	6.5–8.5	3	0.103
TDS	mg/L	500	5	0.172
Ca^{2+}	mg/L	75	2	0.069
Mg^{2+}	mg/L	50	2	0.069
Na^+	mg/L	200	3	0.103
K^+	mg/L	12	1	0.034
HCO_3^-	mg/L	120	2	0.069
SO_4^{2-}	mg/L	250	3	0.103
Cl^-	mg/L	250	3	0.103
NO_3^-	mg/L	45	5	0.172
Total			29	1.000

The normalized weight of each parameter is obtained as:

$$w_i = W_i / \sum_{i=1}^{n} W_i \tag{2}$$

where w_i is the normalized weight for parameter i, W_i is the assigned weight for parameter i, and n is the total number of parameters. Water quality observations are then standardized by dividing the values by their corresponding water quality standard:

$$q_i = C_i / S_i \tag{3}$$

where q_i is the partial WQI score for parameter i, C_i is the observed concentration for parameter i (pH–7 for acidity), and S_i is the water quality standard for parameter i. For the water quality standards, we use the WHO 2011 [26] guidelines for drinking-water, as shown in Table 1. The overall WQI score is obtained by summation of the scores of each parameter multiplied by their normalized weight:

$$\text{WQI} = \sum_{i=1}^{n} w_i q_i \tag{4}$$

A WQI value less than one indicates that the water can be used without any precaution, while higher values indicate poorer water quality. WQI scores make it possible to classify the suitability of water for human consumption into categories such as excellent, good, poor, very poor and unsuitable for use as shown in Table 2 [9,14,15]. Our method for deriving WQI scores does not differ from what is in the literature, the only minor difference is that we have not multiplied the values by one hundred as is traditionally done, but that actually serves no purpose.

Table 2. Classification of the water quality according to the WQI score.

WQI Range	Water Quality Class
<0.5	Excellent
0.5–1	Good
1–2	Poor
2–3	Very poor
>3	Unsuitable

2.4. Geostatistical Data Analysis

Mapping of the WQI estimates is achieved by geostatistical analysis in which an empirical variogram is derived, fitted by a variogram model, cross-validation of the model variogram and regression-kriging using the *gstat* library in R [27,28]. The spatial variation of WQI is modeled as:

$$\mathrm{WQI}(\mathbf{x}) = m(\mathbf{x}) + \epsilon(\mathbf{x}) \tag{5}$$

where $\mathbf{x}$ is the location vector, $m(\mathbf{x})$ is the deterministic component of the water quality index and $\epsilon(\mathbf{x})$ is the stochastic residual. The empirical variogram of the stochastic residual is estimated based on the observations as

$$\gamma(h) = \frac{1}{2N(h)} \sum_{(i,j)\in N(h)} \left[\epsilon(\mathbf{x}_i) - \epsilon\left(\mathbf{x}_j\right)\right]^2 \tag{6}$$

where γ is the variogram or semi-variance, h is the lag distance and $N(h)$ is the number of observed pairs within the lag distance interval. The empirical variogram is fitted by an exponential model:

$$\gamma(h) = n + (s - n)[1 - \exp(-3h/r)] \tag{7}$$

where n is the nugget, s is the sill and r is the practical range. The variogram model is verified by cross-validation whereby the data points are removed one by one and predicted by regression-kriging using the remaining data. Finally, maps of WQI are obtained by spatial interpolation using regression-kriging

$$\mathrm{WQI}(\mathbf{x}) = \sum\nolimits_{k=1}^{p} \beta_k q_k(\mathbf{x}) + \sum\nolimits_{i=1}^{N(\mathbf{x})} \lambda_i \epsilon(\mathbf{x}_i) \tag{8}$$

where β_k are optimized deterministic model coefficients, $q_k(\mathbf{x})$ are external drift components, p is the number of external drift components, λ_i are optimized kriging weights and $N(\mathbf{x})$ is the number of data points selected within the neighborhood of $\mathbf{x}$.

3. Results and Discussion

3.1. Hydro-Chemical Characterization

The results of in the in-situ tests and laboratory analyzes of the water samples from the 19 municipal wells in the Dier al-Balah Governorate in 2009 and 2014 are shown in Table 3; Table 4, respectively. These tables also provide the average and standard deviation for each parameter. The second-to-last column of Tables 3 and 4 shows the charge-balance error obtained with Equation (1). All charge-balance errors are within an acceptable limit of ±10%, except for the sample from well W11 in 2014, which is just above the limit. This small might be due to some ions missing in the analysis, such as for example organic acids that may be present due to contamination by industrial or household waste.

Table 3. Physio-chemical analysis of the groundwater samples taken in 2009; also given are the charge-balance error *E*, the WQI score, the water quality classification, and the mean and standard deviation (SD).

Well	pH	TDS	Ca^{2+}	Mg^{2+}	Na^+	K^+	HCO_3^-	SO_4^{2-}	Cl^-	NO_3^-	E	WQI	Water Quality
ID		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	%	Score	Class
W1	7.40	3241	190	105	670	5.7	273	265	1126	139	3.76	3.15	Unsuitable
W2	7.10	2737	161	103	480	7.9	574	267	920	200	−7.97	3.18	Unsuitable
W3	7.30	2912	182	132	524	8.1	421	252	987	191	−0.14	3.22	Unsuitable
W4	7.60	3493	226	125	640	14.5	323	204	1269	183	1.49	3.54	Unsuitable
W5	7.10	3108	160	141	600	7.5	268	380	1140	147	−0.99	3.18	Unsuitable
W6	7.60	2877	131	86	775	5.5	231	272	1028	54	9.34	2.62	Very poor
W7	7.00	2779	174	128	620	5.1	286	241	973	113	8.64	2.82	Very poor
W8	7.10	2597	160	139	660	9.1	370	511	821	132	7.04	2.96	Very poor
W9	7.90	1953	86	57	480	4.0	233	204	662	38	4.54	1.82	Poor
W10	7.80	2198	111	71	520	4.2	246	209	759	62	5.11	2.11	Very poor
W11	8.00	715	62	31	130	1.2	201	60	172	50	5.43	0.92	Good
W12	7.60	1932	112	58	460	11.1	323	186	655	99	2.33	2.12	Very poor
W13	7.50	2114	80	35	440	2.6	266	186	717	61	−6.02	1.94	Poor
W14	7.30	1631	84	38	310	2.8	360	65	552	42	−5.96	1.57	Poor
W15	7.60	2002	74	53	400	4.1	254	227	632	43	−3.53	1.81	Poor
W16	7.20	3598	165	123	780	17.4	427	511	1160	190	−0.64	3.77	Unsuitable
W17	7.50	2100	138	73	360	6.7	248	128	681	102	2.05	2.12	Very poor
W18	7.50	2233	121	60	455	5.1	273	156	681	120	3.41	2.27	Very poor
W19	7.50	2436	150	97	370	7.7	278	407	698	120	−4.37	2.49	Very poor
Mean	7.45	2456	135	87	509	6.9	308	249	823	110	1.24	2.51	
SD	0.28	700	45	37	163	4.0	90	126	265	55	5.08	0.75	

Table 4. Physio-chemical analysis of the groundwater samples taken in 2014; also given are the charge-balance error *E*, the WQI score, the water quality classification, and the mean and standard deviation (SD).

Well	pH	TDS	Ca^{2+}	Mg^{2+}	Na^+	K^+	HCO_3^-	SO_4^{2-}	Cl^-	NO_3^-	E	WQI	Water Quality
ID		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	%	Score	Class
W1	7.95	3409	207	117	760	6.0	328	312	1279	144	2.80	3.43	Unsuitable
W2	7.76	2912	176	116	560	6.3	598	331	1032	211	−6.91	3.44	Unsuitable
W3	7.50	3493	198	147	660	6.8	578	353	1179	236	−2.89	3.91	Unsuitable
W4	7.65	3514	243	168	760	15.2	397	273	1389	233	3.73	4.01	Unsuitable
W5	7.54	3395	179	162	740	7.7	281	413	1272	171	2.64	3.57	Unsuitable
W6	7.71	3031	155	98	863	5.6	271	329	1259	72	5.44	2.97	Very poor
W7	7.53	2968	185	137	674	5.7	313	273	1124	122	5.80	3.06	Unsuitable
W8	8.16	3003	189	146	720	5.4	409	535	1022	169	3.43	3.42	Unsuitable
W9	7.77	1862	96	59	509	4.2	246	241	708	47	3.43	1.90	Poor
W10	7.51	2359	132	86	634	5.7	323	277	853	79	6.42	2.45	Very poor
W11	7.66	774.2	64	44	164	1.4	216	71	191	62	10.16	1.04	Poor
W12	7.61	2170	136	74	503	6.1	382	262	794	118	−1.56	2.45	Very poor
W13	8.20	2464	98	53	523	3.7	304	218	893	83	−5.80	2.35	Very poor
W14	8.10	1729	97	58	398	3.7	394	103	623	52	0.01	1.81	Poor
W15	7.09	2366	91	67	530	4.5	360	277	777	55	−1.84	2.22	Very poor
W16	7.90	3906	169	136	860	20.0	456	554	1326	204	−1.84	4.11	Unsuitable
W17	8.00	2954	153	92	461	7.5	293	154	998	126	−3.72	2.77	Very poor
W18	8.10	2975	133	83	630	6.1	311	182	977	133	3.06	2.87	Very poor
W19	7.68	2618	169	106	503	6.8	311	442	872	132	−2.26	2.81	Very poor
Mean	7.80	2732	151	103	603	6.8	356	295	977	129	1.06	2.87	
SD	0.30	749	47	39	170	4.2	101	129	293	62	4.51	0.81	

Tables 3 and 4 indicate that the ionic dominance pattern is of the order $Na^+ > Ca^{2+} > Mg^{2+} > K^+$ for the cations and $Cl^- > HCO_3^- > SO_4^{2-} > NO_3^-$ for the anions, both in 2009 and 2014. Figure 2 shows the major ion composition in a piper plot. All water quality samples are clustered in the sodium-chloride category ($Na^+ + K^+ > 50\%$, $Cl^- > 50\%$), from which we can conclude that the groundwater is very brackish. Also shown in Figure 2 is the theoretical mixing line between fresh groundwater (100% $Ca^{2+} + Mg^{2+}$, 100% HCO_3^-) and Eastern Mediterranean seawater (78% Na^+, 100% $Cl^- + SO_4^{2-}$) [29].

All groundwater samples are clustered around this line, which suggest that the groundwater quality is likely to be influenced by mixing with seawater.

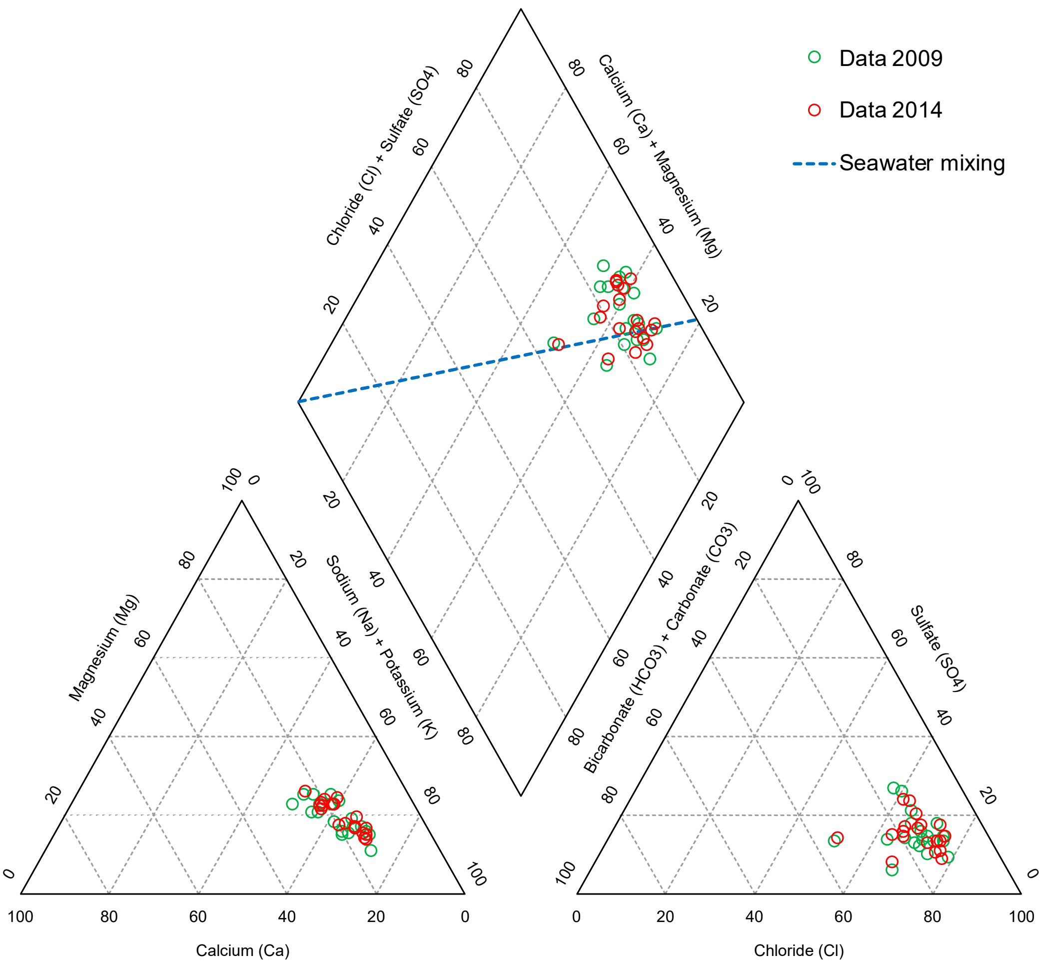


Figure 2. Piper plot showing the major ion composition of the groundwater samples and the theoretical mixing line between fresh groundwater and seawater.

There appears to be little difference in the major ion composition between the groundwater samples collected in 2009 and 2014. However, the Piper diagram shows only the relative distribution, while absolute concentration values indicate a clear increase in salinity over time. This is shown in Figure 3 where the total cation concentration is plotted against the chloride concentration, both expressed in milliequivalent per liter. The total cation content is used here because the relative distribution of cations is influenced by adsorption and desorption of cations on soil particles, while chloride is a conservative tracer. In the graph, samples taken from a particular well in 2009 and 2014 are linked to show the chemical evolution over time, which clearly indicates that for all wells both the total cation content and chloride concentration increase considerably. The solid blue line in the graph shows the chemical evolution of mixing fresh water with East-Mediterranean seawater; the ratio between the total cation content and the chloride content for Eastern Mediterranean seawater is 1:0.90 [29]. All the groundwater samples plot above this line because groundwater has a larger cation versus chloride ratio than East-Mediterranean seawater due to the presence of more bicarbonate and sulfate. However, the increase in ion content of the well samples from 2009 to 2014 is generally parallel to the sea water mixing line, which suggests that the groundwater is constantly mixed with seawater. A probable explanation for the chemical evolution of the groundwater is given by the dotted line, which shows the mixing of seawater with groundwater with an initial ion content of 20 meq/L without chloride.

This line runs more or less through most of groundwater samples and indicates that the chemical evolution of the groundwater is strongly determined by the intrusion of seawater. The marks on this mixing line indicate increases by 1% sea water during the mixing process. Most groundwater samples plot along a range of 3–6% of seawater mixing, while the increase in ion content from 2009 to 2014 is approximately 1% of seawater mixing.

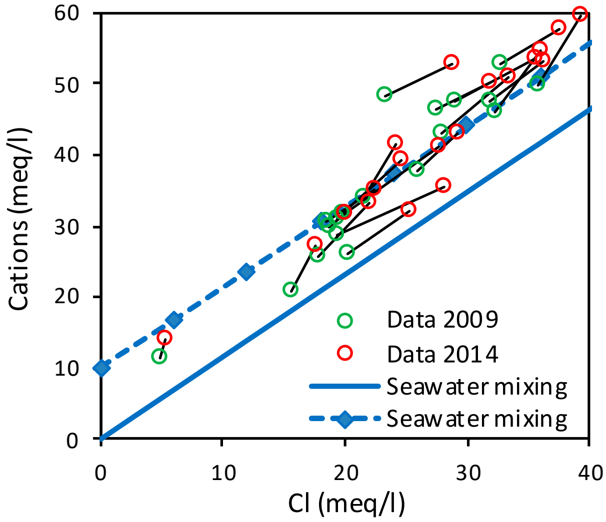


Figure 3. Plot of the total cation concentration versus the chloride concentration and the theoretical mixing line between fresh groundwater and seawater (solid line) or between groundwater with an initial ion content of 20 meq/L and no chloride and seawater (dotted line, the markings indicate intervals of 1% increase in sea water mixing).

3.2. Water Quality

The pH-values show that all samples have alkaline properties as expected for groundwater. There seems to be a slight increase from 2009 to 2014, but all values are well within the WHO standard. Although the pH usually has no direct influence on human health [26], it can have an effect on the hardness of water. There are also no real health guidelines for major ions or TDS in drinking water, but excessive salinity can have an effect on odor and taste, while high levels of calcium and magnesium can lead to scaling. All TDS values are well above the WHO standard, some are 5 to 6 times larger, making the water less suitable for domestic use. TDS increased slightly from 2009 to 2014, except for wells W17 and W18 for which TDS increased by approximately 30%; these wells are close to the coastline. TDS values for groundwater that exceed the WHO standard in Gaza have been reported by other researchers; Aish [8] reported TDS values between 680 mg/L and 3107 mg/L for 9 municipal wells sampled in 2009 and Abbas et al. [9] reported that 88% of the TDS values sampled from 58 municipal wells in 2010 exceeded the WHO standard.

For the major ions, excessive amounts of sodium, chloride and bicarbonate are found, two to four times higher than the WHO standards and for calcium and magnesium one to two times highr than the WHO standards in all wells, except well W11. The levels of potassium and sulfate are moderate and are all below or around the WHO standards. The major ion levels in all wells increased from 2009 to 2014 except potassium. In particular, chloride and sodium levels increased significantly in all wells, some by more than 30%, suggesting salinization due to seawater intrusion in the aquifer. Excessive concentrations of sodium and chloride above WHO standards in the Gaza Strip have also been reported by other researchers; Aish [8] reported sodium levels in the range of 109–680 mg/L and chloride levels in the range 179–1231 mg/L for nine municipal wells sampled in 2009, Abbas et al. [9] reported that 67% of the 58 wells sampled in 2010 had sodium levels above the WHO standard and 71% above the WHO standard for chloride, Abu-alnaeem et al. [12] found that 75% of the 219 wells sampled in 2013–2014 were above the WHO standard for sodium and 80% above the WHO standard for chloride.

High levels of nitrate in drinking water are undesirable because of a potential reduction to nitrite that can be hazardous to health, especially for infants and pregnant women. Moreover, nitrate in groundwater is mainly due to contamination, for example leaching of fertilizers or manure, wastewater disposal, leakage from septic tanks, etc., and as such it indicates the possible presence of other hazardous contaminants, such as bacteria or pesticides. Almost all nitrate levels in the well samples are above the WHO standard, some even three to four times higher; only the samples from wells W9, W14 and W15 in 2009 had a nitrate concentration within the permissible range. Large-scale pollution of groundwater by nitrate has also been reported in other studies; Shomar [4] found that 89% of the 94 wells sampled in 2002–2004 showed nitrate levels above the WHO standard, Almasri and Ghabayen [7] reported that 75% of the 2413 nitrate concentrations observed in 568 wells in 1990 and 2000–2004 exceeded the WHO standard and Abu-alnaeem et al. [12] reported that 90% of 219 wells were nitrate polluted in 2013–2014.

The WQI scores and associated water quality classes are given in the last columns of Tables 3 and 4. All WQI scores are all larger than one, except well W11 in 2009, which indicates that the water quality is problematic. The average WQI score was 2.51 in 2009 and rose to 2.87 in 2014, demonstrating a continuous deterioration in water quality. All water samples collected from wells close to the coast have WQI scores greater than two, indicating very poor water quality and about half of them even have values greater than three, indicating that the groundwater is unsuitable for human consumption. The inland wells generally have WQI scores less than 2.5 and can be classified as having poor or very poor water quality, with the exception of well W11 in 2009, the only groundwater sample with good quality.

3.3. Water Quality Mapping

The water quality of the well samples is clearly influenced by the proximity to the coastline, which clearly indicates a strong influence of salinization due to sea water intrusion. Figure 4 shows the observed WQI scores plotted against the distance to the coast. Also shown in the figure are the fitted linear regression lines through the data points, which suggest a significant decrease in WQI score with distance to the sea. The influence of the proximity of the coastline was also noted by other researchers [6,9,12].

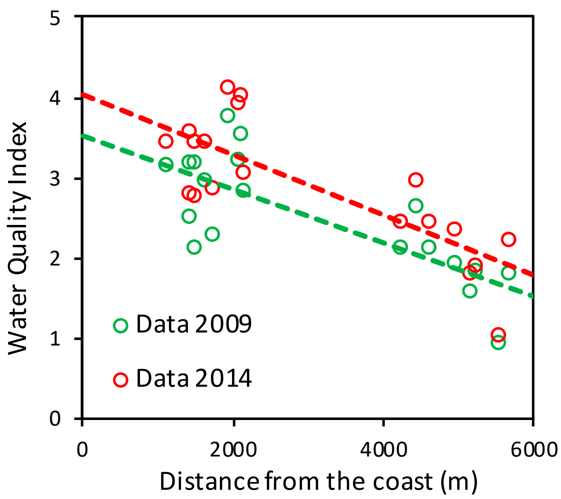


Figure 4. Observed water quality index plotted against the distance to the coastline and the fitted linear regression lines.

The distance to the coast is therefore an external drift component that explains the deterministic part of the spatial model for WQI, expressed by Equation (5). The stochastic residuals of the model can be estimated from Figure 4 as the differences between the WQI values and the corresponding values derived by linear regression. These residuals make it possible to determine the experimental variogram with Equation (6), for which we used a lag of 500 m. The results are shown in Figure 5, which indicates a spatial correlation of the residuals up to a distance of about 3000 m. The empirical variograms of the data for 2009 and 2014 do not differ much, so one suitable variogram model can be fitted to all the data with the following parameters: exponential type, zero nugget, a sill of 0.23 and a practical range of 3000 m. This variogram model is verified by cross-validation using regression-kriging with the distance to the coastline as a drift component. The results are given in Figure 6, which shows the estimated WQI scores versus the observations. The 1:1 identity line and the 95% confidence bands are also shown. The latter are derived as ±1.96 times the standard deviation of the residuals, which can be estimated as the square root of the sill. The results show that all estimated values are within acceptable range, which proves that the variogram model is acceptable.

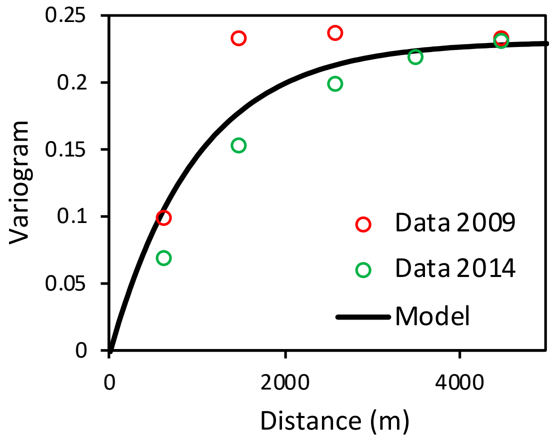


Figure 5. Empirical variogram of the stochastic residual of the water quality index and fitted exponential variogram model.

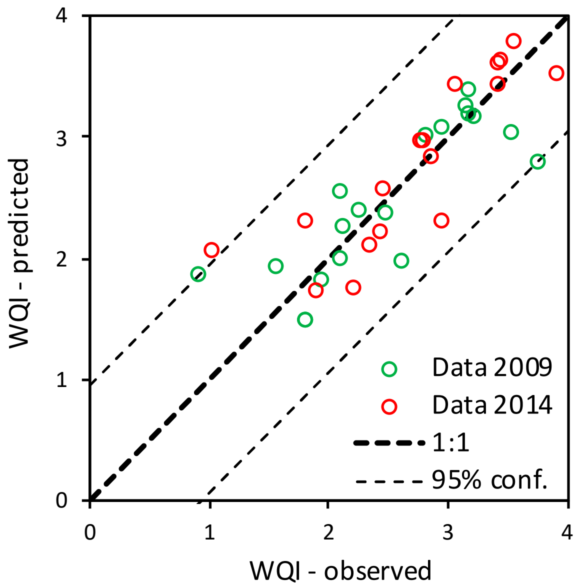


Figure 6. Cross-validation results with estimated WQI scores versus observations and the 1:1 identity line with 95% confidence bands (conf.).

Water quality index maps for 2009 and 2014 derived by regression-kriging are shown in Figure 7. The maps clearly indicate a regional trend in water quality as a result of sea water intrusion, in addition to some small-scale variations as a result of local contamination by domestic, agricultural or industrial waste. The regional trend shows a decrease in WQI by about 0.4 per km distance from the sea. Local increases in WQI scores are noted around well W16 and around wells W3 and W4, the first due to high sulfate levels and the later due to high levels of bicarbonate. Comparison of the maps of 2009 and 2014 shows a clear increase of the WQI scores over time by about 0.36 on average or about 0.07 per year, especially in the regional trend that expresses the intrusion of salt water. Qahman and Larabi [30] also reported a continuous decline in groundwater quality as a result of over-exploitation of groundwater resulting in seawater intrusion.

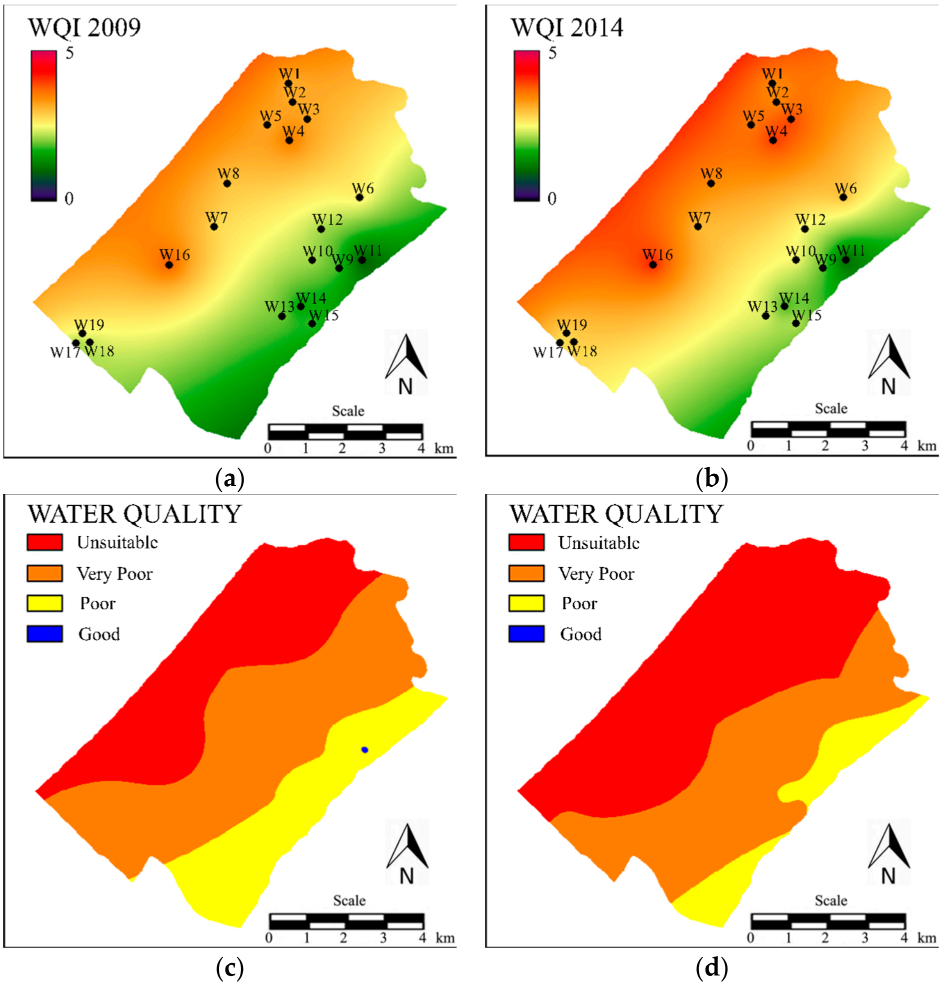


Figure 7. Maps with the spatial distribution of (**a**) WQI scores based on data from 2009, (**b**) WQI scores based on data from 2004, (**c**) groundwater quality classification for 2009 and (**d**) groundwater quality classification for 2014.

The groundwater quality classification maps for 2009 and 2014 are also shown in Figure 7. For both years there are only three important water quality classes: poor, very poor and unsuitable for human consumption. In 2009, about 35% of the groundwater in the governorate was unsuitable for human consumption, which increased to 53% in 2014. All unsuitable groundwater is along the coast and the figures show that the boundary between unsuitable and very poor groundwater quality shifted by around 700 m inland in a span of 5 years. If this trend continues, there will soon be no safe groundwater left.

4. Conclusions

Water samples collected from 19 municipal wells in April 2009 and April 2014 in the Dier al-Balah Governorate of the Gaza Strip indicate a sodium-chloride dominated water quality. Plotted in a piper diagram, the major ion composition appears to be clearly influenced by mixing with Mediterranean seawater. When the chloride concentration is plotted against the total cation concentration, it can also be concluded that the observed groundwater samples are mainly the result of mixing with 3–6% seawater, while in the period 2009–2014 there was an increase by about 1% more seawater mixing with the groundwater.

Almost all observed chemical parameters are above their WHO water quality standard, in particular for total dissolved solids, sodium, chloride and nitrate and for the wells close to the coastline. The differences between 2009 and 2004 show an increase in all major ion concentrations in particular for sodium and chloride. The resulting scores for the water quality index indicate that the water quality is very problematic, since almost all samples classify from poor water quality to unsuitable for human consumption.

The observed water quality depends on the distance to the coastline. Thus, the spatial variation of the WQI scores can be described by a model with a deterministic drift depending on the distance to the coastline and a stochastic residual described by an exponential variogram and a practical range of 3000 m. This model is verified by cross-validation and mapping of the groundwater quality is achieved by regression-kriging. The results indicate that the groundwater in a large area along the coastline is not suitable for human consumption and that this zone considerably expanded inland over a period of five years (2009–2014). It is therefore appropriate that the authorities concerned in the Dier al-Balah Governorate closely monitor the continued deterioration of groundwater, take the necessary precautions to prevent use of contaminated water for human consumption and develop alternatives or corrective measures to meet the water needs of the region.

Seawater intrusion in the Gaza Strip is a national problem, but mitigation measures are lacking. Possible solutions and mitigation measures as rain water harvesting, seawater desalination and wastewater treatment and reuse are well known and have been recommended in the literature (e.g., [31,32]). It is recommended that seawater intrusion be regularly monitored to improve groundwater management in the Gaza strip.

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