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ORIGINAL ARTICLE

Salinization of groundwater in arid and semi-arid zones: an example from Tajarak, western Iran

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Abstract Study of the groundwater samples from Tajarak area, western Iran, was carried out in order to assess their chemical compositions and suitability for agricultural purposes. All of the groundwaters are grouped into two categories: relatively low mineralized of Ca–HCO_3 and Na–HCO_3 types and high mineralized waters of Na–SO_4 and Na–Cl types. The chemical evolution of groundwater is primarily controlled by water–rock interactions mainly weathering of aluminosilicates, dissolution of carbonate minerals and cation exchange reactions. Calculated values of pCO_2 for the groundwater samples range from 2.34×10^{-4} to 1.07×10^{-1} with a mean value of 1.41×10^{-2} (atm), which is above the pCO_2 of the earth's atmosphere ($10^{-3.5}$). The groundwater is oversaturated with respect to calcite, aragonite and dolomite and undersaturated with respect to gypsum, anhydrite and halite. According to the EC and SAR the most dominant classes (C3-S1, C4-S1 and C4-S2) were found. With respect to adjusted SAR (adj SAR), the sodium (Na^+) content in 90% of water samples in group A is regarded as low and can be used for irrigation in almost all soils with little danger of the development of harmful levels of exchangeable Na^+, while in 40 and 37% of water samples in group B the intensity of problem is moderate and high, respectively. Such water, when used for irrigation will lead to cation exchange and Na^+ is adsorbed on clay minerals while calcium (Ca^{2+}) and magnesium (Mg^{2+}) are released to the liquid phase. The salinity hazard is regarded as medium to high and special management for salinity control is required. Thus, the water quality for irrigation is low, providing the necessary drainage to avoid the build-up of toxic salt concentrations.

Keywords Salinization · Groundwater · Irrigation · Iran · Arid and semi-arid

Introduction

Salinity and sodicity are the principal water quality concerns in irrigated areas of arid and semi-arid regions using poor water quality for irrigation. Irrigation water quality has a significant role in crop production and has a profound impact on physical and chemical soil properties. In arid and semi-arid regions, agriculture is mainly limited by the availability of suitable irrigation water, and groundwater is the main source of irrigation. However, even with sufficient water, its use is often not suitable, leading to soil salinization as a consequence of inappropriate irrigation and drainage techniques (Luedeling et al. 2005). Use of poor quality groundwater has become inevitable for irrigation to compensate rapidly increasing water demands in many arid and semi-arid regions. The greatest threat to maintaining freshwater supplies is the depletion of the surface and groundwater resources that are used to fulfil the needs of the rapidly growing human population. This has been a crucial factor to agricultural production in many arid and semi-arid regions (Bouwer 2000). Current supplies of good-quality surface and groundwaters for crop production have not been able to keep pace with rapidly increasing water demands as a consequence of increased cropping

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intensities (Pimental et al. 1999) and/or expansion in irrigated agriculture on marginal lands (Qadir et al. 2001). Groundwaters of different qualities are being used to make up the shortage of good-quality waters for crop production. This may lead to soil and drainage water contamination by a variety of pollutants. Irrigated agriculture in semi-arid and arid areas should take into account the risk of progressive salinization of soils, due to solute accumulation from irrigation water. Salinity and sodicity are the principal water quality concerns in irrigated areas receiving such water (Ayars and Tanji 1999). Increasing use of low-quality water for irrigation requires that we evaluate the suitability for irrigation. An understanding of the quality of water used for irrigation and its potential negative impacts on crop growth is essential to avoid problems and to minimize production. However, the evaluation and management of groundwater resources require an understanding of hydrogeological and hydrochemical properties of the aquifer (Umar et al. 2001). In Tajarak, a part of Hamadan province, the aquifers are of major importance as they are the main source of water supply. Hydrogeochemical investigation was carried out in Tajarak area in Hamadan province, western Iran, to assess chemical composition of groundwater and its suitability for irrigation purposes.

Materials and methods

Study area

The study area is in the northern region of Tajara area, Hamadan province, covering an area of about 1,236 km^2 and located between longitudes 48°50′38″ and 49°26′42″E and latitudes 34°36′56″ and 35°12′47″N (Fig. 1). The average elevation (altitude) for the study area is 1,650 m.a.s.l. The area has a cold temperate climate, with mean maximum summer temperatures (July) about 23.50 and minimum winter temperatures (January) of –1.90. The climate of the study area is considered to be semi-arid, the annual precipitation being approximately 300 mm. Late spring and summer months were driest, each with monthly rainfall below 5 mm (Sabziparvar 2003). The annual potential evaporation, which far exceeds the annual rainfall with a mean annual amount approximately estimated from 1975 to 2001, is 1,505 mm (Sabziparvar 2003). High evapo-ratranspiration rates result in the deterioration of groundwater quality over time. The most important economic activity of the area is agriculture, and the chief crops are garlic, potato and wheat, with actual irrigation being lower than total theoretical demand, as there is considerable deficit in relation to the amount of irrigated land.

The Hamadan area is characterized by the predominance of metamorphic rocks of both sedimentary and magmatic origins (Sepahi 1999). The metamorphic rocks constitute an assemblage of high to low metamorphic grade issued from sedimentary sequences that have been affected by a more or less developed tectonometamorphic event (Baharifar et al. 2004).

The aquifer is heterogeneous in regard to the spatial distribution of hydraulic conductivity. Aquifer parameter in Hamadan plain for hydraulic conductivity is 1.3–46 m day^{-1}. Typically, recharge to regional flow systems occurs as snow melt in high elevations; discharge occurs in low basins, which receive less annual precipitation than the higher elevations. Other source of recharge is rainfall, floodflows, the subsurface

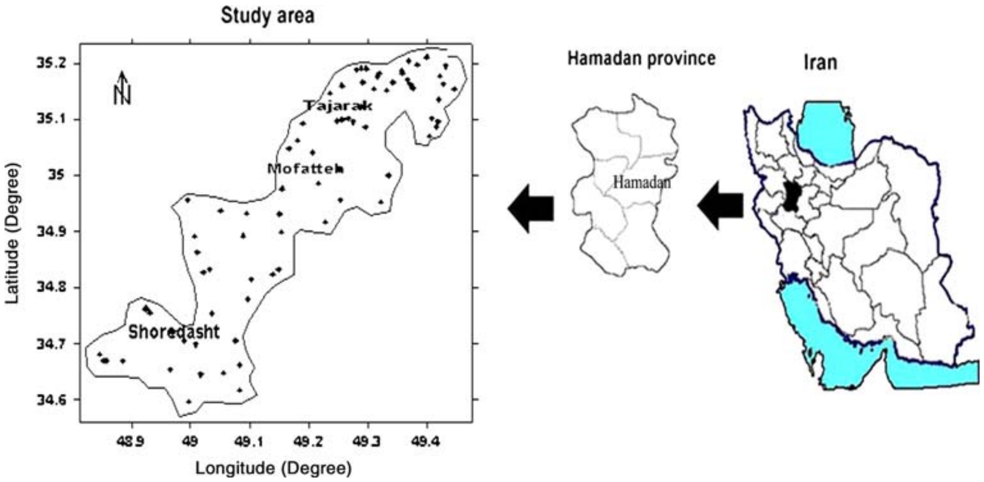


Fig. 1 Study area location

seepage from irrigation and drainage canals and direct infiltration of return flow water irrigation. The transmissivity and storativity have been estimated by pumping tests in seven points of the plain and vary between 127–4,654 m^2 day^{-1} and 1–6.5%, respectively (Marofi 2003).

Most soils are highly calcareous and are mostly classified as Aridisol. Some of the relevant chemical and physical properties of the soil samples from the studied area are as follow: clay contents ranged from 135 to 431 g kg^{-1}, cation exchange capacity ranged from 11.3 to 18 $cmol_c$ kg^{-1}, and organic carbon contents were low in soils ranging from 3.2 to 37 g kg^{-1}. The soils are sandy loam to clay in texture. The clay fraction contained mainly illite, smectite, chlorite and vermiculite (Jalali 2006a).

Water samples

Eighty groundwater samples were collected from Tajarak area, during 2000 (Fig. 1). Samples were analysed in the laboratory for the major ion chemistry employing standard method. The determinations were made within 48 h after collection. The pH and electrical conductivity (EC) were measured using pH and EC meters. Calcium and Mg^{2+} were determined titrimetrically using standard EDTA. Chloride (Cl^-) was determined by standard $AgNO_3$ titration. Carbonate (CO_3^{2-}) and bicarbonate (HCO_3^-) were determined by titration with HCl. Sodium and potassium (K^+) were measured by flame photometry. Sulphate (SO_4^{2-}) by spectrophotometer turbidimetry and nitrate (NO_3^-) by colorimetry with an UV-visible spectrophotometer (Rowell 1994). Total dissolved solids (TDS) were computed by multiplying the EC (dS m^{-1}) by a factor of 640. Care was taken that the HCO_3^- and Ca^{2+} ions were analysed within 24 h of sampling.

Results and discussion

Major ion chemistry

The calculated mean and standard deviations of the major ion concentrations, the pH, pCO_2 and saturation indices (SI) for anhydrite, aragonite, calcite, dolomite, gypsum and halite for the groundwater in the study area are listed in Table 1. Also the distribution pattern of the concentration of different parameters is shown as contour maps (Figs. 2, 3). Spatial variation in the chemical parameters of the groundwater reflects variations in natural and human activities. Among the cations, the concentrations of Na^+, Ca^{2+}, Mg^{2+} and K^+

Table 1 Chemical compositions (major elements) in the well water samples

	Ca^{2+}	Mg^{2+}	K^+	Na^+	Cl^-	CO_3^{2-}	HCO_3^-	NO_3^-	SO_4^{2-}	EC	pH	TDS	SI (gypsum)	SI (dolomite)	SI (calcite)	SI (anhydrite)	SI (aragonite)	SI (halite)	Log pCO_2
Mean	3.07	2.68	0.24	10.79	7.74	0.22	5.41	0.95	3.99	2.17	7.66	1,394	−1.45	1.01	0.49	−1.66	0.36	−6.18	1.41E-02
Max	11.3	9.25	4.54	63.13	54	1.5	11.7	4.07	20	8.9	8.76	5,696	−0.18	2.97	1. 54	−0.38	1.41	−4.43	1.07E-01
Min	1.0	0.35	0.00	0.35	0.4	0.0	0.8	0.12	0.1	0.39	6.5	249	−2.85	−1.18	−0. 49	−3.04	−0.60	−8.16	2.34E-04
SD	1.98	1.97	0.67	11.45	8.33	0.32	2.43	0.77	4.81	1.60	0.51	1,030	0.60	0. 85	0.39	0.62	0.38	0.79	2.05E-02

Concentrations are expressed in mmol l^{-1}, TDS in mg l^{-1}, EC in dS m^{-1}

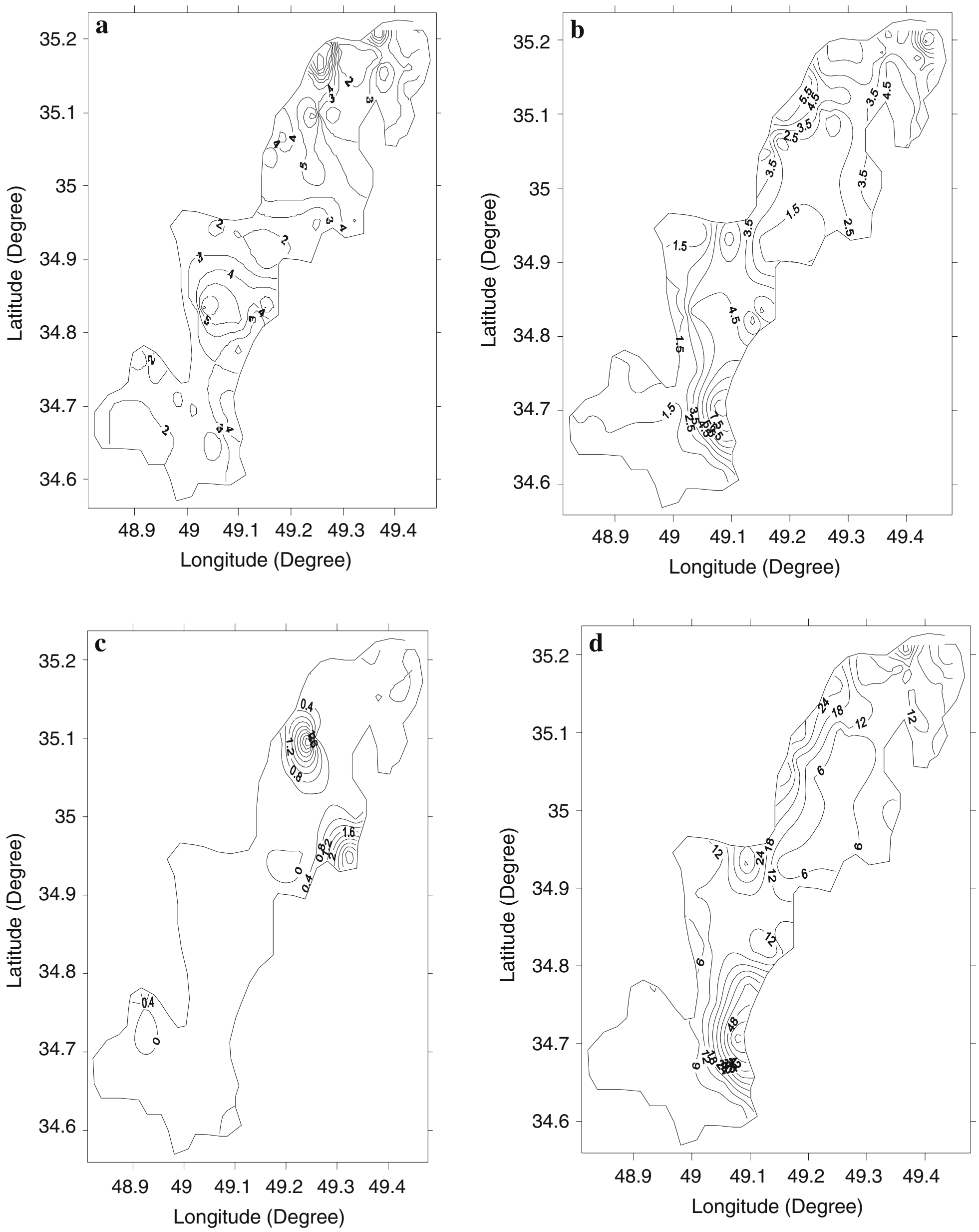


Fig. 2 Spatial distributions of **a** Ca^{2+}, **b** Mg^{2+}, **c** K^{+} and **d** Na^{+} (mmol l^{-1}) in the groundwater in the study area

ions ranged from 0.35 to 63, 1–11.3, 0.35–9.2 and 0.006–4.5 mmol l^{-1} with a mean of 10.8, 3, 2.7 and 0.25 mmol l^{-1}, respectively. Their concentrations (on the basis of meq l^{-1}) represent on average 41.1, 32.6, 25.2 and 1.1% of all the cations, respectively. Thus, the order of abundance is $Na^{+} > Ca^{2+} > Mg^{2+} > K^{+}$. Major anions show a very wide range of variations. Among the anions, the concentrations of HCO_3^-, Cl^-, SO_4^{2-}, NO_3^- and CO_3^{2-} ions lie in between 0.8 and 11.7, 0.4 and 54, 0.2 and 40, 0.12 and 4.1 and 0 and 3 meq l^{-1} with a mean of 5.4, 7.7, 4, 0.95 and 0.45 meq l^{-1}, respectively. The order of their abundance is $HCO_3^- > Cl^- > SO_4^{2-} > NO_3^- > CO_3^{2-}$, contributing on average (meq l^{-1}), 31.7, 31.5, 28.1, 5.7 and 3% of the total anions, respectively. The EC indicates the amount of material dissolved in water and its values ranges from 0.39 to 8.9 with a mean of 2.2 dS m^{-1}. TDS ranges from 250 to 5,696 with an average of 1,395 mg l^{-1}. Average pH is 7.7, the maximum is 8.8 and the minimum is 6.5.

Hydrochemical facies

The Piper diagram in Fig. 4 constructed using Aquchem Scientific Software shows the relative concentrations of the different ions from individual water samples. Nine groundwater groups have been identified on the basis of major ion concentrations, namely:

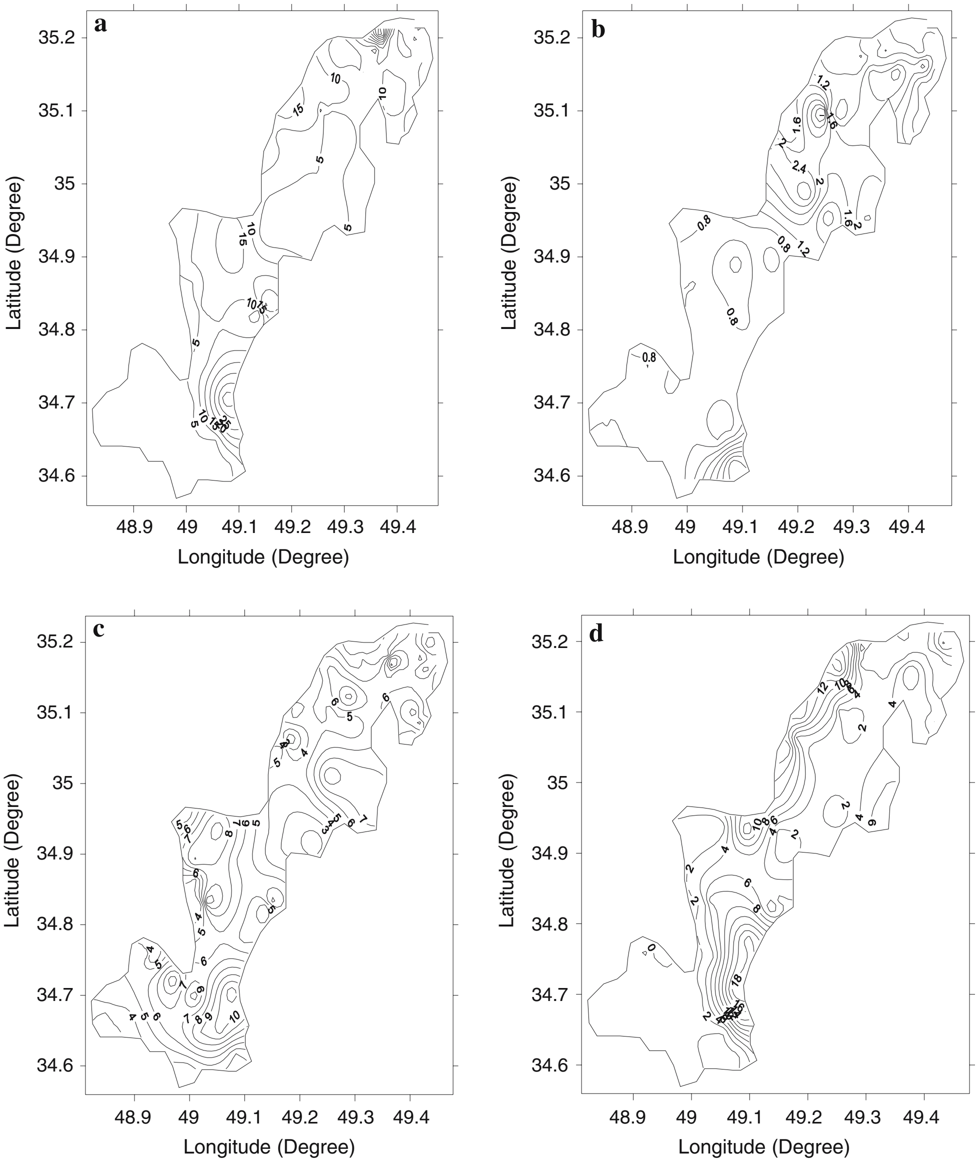


Fig. 3 Spatial distributions of **a** Cl^-, **b** NO_3^-, **c** HCO_3^- and **d** SO_4^{2-}(mmol l^{-1}) in the groundwater in the study area

(1) Na–Ca–HCO_3–Cl, Na–Ca–HCO_3–SO_4–Cl, Na–HCO_3–Cl.
(2) Na–Mg–SO_4–Cl, Na–Ca–Mg–SO_4, Na–Ca–Mg–SO_4–HCO_3, Na–Ca–Mg–SO_4–HCO_3.
(3) Ca–Mg–HCO_3, Ca–Mg–HCO_3–SO_4, Ca–Mg–HCO_3–NO_3, Ca–Mg–HCO_3–Cl, Ca–Mg–Na–HCO_3–SO_4.
(4) Na–Ca–Cl–HCO_3, Na–Ca–Cl–SO_4–HCO_3, Na–Ca–Mg–Cl–HCO_3–SO_4.
(5) Mg–Ca–HCO_3, Mg–Ca–Na–HCO_3–SO_4, Mg–Na–HCO_3–SO_4, Mg–Ca–HCO_3–NO3– CO3, Mg–Ca–HCO_3–SO_4–Cl.
(6) Ca–Mg–SO_4–HCO_3, Ca–Na–Mg–SO_4–HCO_3, Ca–Mg–Na–SO_4–HCO_3, Ca–Mg–SO_4– HCO_3.
(7) Ca–Mg–Cl–HCO_3–SO_4, Ca–Mg–Cl–SO_4, Ca–Mg–Cl–HCO_3, Ca–Na–Mg–Cl–HCO_3.
(8) Mg–Ca–Na–SO_4, Mg–SO_4–HCO_3–Cl, Mg–Ca–SO_4–HCO_3, Mg–Ca–Na–SO_4–HCO_3
(9) Mg–Ca–Na–Cl, Mg–Cl–HCO_3, Mg–Ca–Na–Cl–SO_4.

Based on dominant cations and anions these nine groups are shown in the following way: Na–HCO_3, Na–SO_4, Ca–HCO_3, Na–Cl, Mg–HCO_3, Ca–SO_4, Ca–Cl, Mg–SO_4 and Mg–Cl. The average chemical compositions for the eight of these sub-types are given in Table 2.

The Na–SO_4, Na–Cl, Ca–HCO_3 and Na–HCO_3 each represent 25, 20, 17.5 and 12.5% of the total number of water samples analysed, while Ca–SO_4 and Ca–Cl each represent 7.5% of the total number of water samples analysed. The other three water types represent only

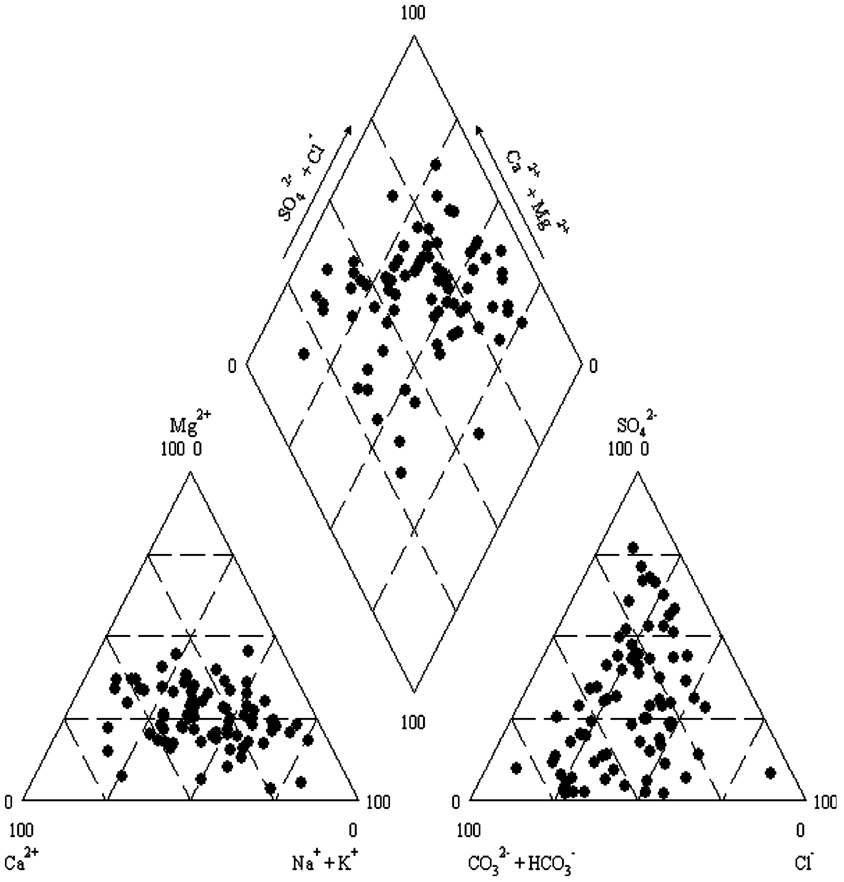


Fig. 4 The piper diagram for the groundwater samples of the study area

10%. The Na–SO_4 type water is dominated in the most part of studied area (Fig. 5) with the Na^+ range from 3.5 to 63 mmol l^{-1}, while SO_4^{2-} range from 1 to 20 mmol l^{-1}. The average TDS value for the Na–SO_4 and Na–Cl water types is 2,277 and 1,727 mg l^{-1}, respectively. The Ca–HCO_3 (average TDS = 648 mg l^{-1}) and Na–HCO_3 (average TDS = 702 mg l^{-1}) type waters are less mineralized than other type waters. In general, a significant increase in the degree of water mineralization in the direction of flow is observed and the least mineralized water was found closest to the main recharge area.

Water groups

The most important parameters used in water type identification are major cations and anions concentrations, TDS, and cations and anions ratios. This method is widely used by other researches to divide chemical data into groups (Hiscock 1993; Tellam 1994; Kimblin 1995; Frapporti et al. 1995; Oetting et al. 1996; Saleh et al. 2001). In general, according to the analytical data, all the groundwater investigated can be classified into two groups. Group A groundwaters are low in mineralization (TDS < 1,000 mg l^{-1}) and is representative of most wells (52% of total wells) throughout the study site. The TDS vary from 250 to 982 (Table 3), the average value being of 653, with low Cl^- and SO_4^{2-} content. Among the major cations, a predominance of Ca^{2+} is observed for this group of groundwaters. Concentrations of Ca^{2+} (in meq/l) represent on average 38% of all the cations. Sodium ions are secondary in important, representing on average 34% of all cations. Among the major anions, HCO_3^- generally dominates, representing on average 42% of all the anions. Chloride, SO_4^{2-} and NO_3^- ions are not abundant; they represent on average 27, 19 and 7.5% of all the anions, respectively. Groundwater in this group is mainly Ca–HCO_3 and Na–HCO_3 facies. The ionic ratios $Ca^{2+}/(Na^+ + K^+)$ and K^+/Na^+ for these groundwaters are very heterogeneous (Fig. 7). Such heterogeneity may be attributed to local variations in the mineralogical nature of the groundwater reservoir and/or to geochemical processes occurring in the aquifer (Njitchoua et al. 1997). The weak salinity of these groundwaters is thought to be linked either to a shorted time for weathering, and/or to the lack of sufficient amounts of soluble amounts of soluble cation bearing minerals in the aquifer rock environment.

Table 2 Range and mean chemical compositions and standard deviation in the 8 sub-types of groundwater samples

Facies	Statistical parameters	Physical parameters		Cations and anions (mg l^{-1})							
		pH	EC (dS m^{-1})	K^+	Na^+	Ca^{2+}	Mg^{2+}	Cl^-	NO_3^-	SO_4^{2-}	HCO_3^-
Ca–HCO$_3$	Maximum	8.4	2.15	38.86	140.3	224	55.2	177.5	97.87	288	616.1
	Minimum	6.5	0.39	0.22	8.05	46	16.8	14.2	15.05	9.6	152.5
	Mean	7.54	1.01	4.53	54.96	98.14	33.94	83.67	44.63	82.62	305.50
	SD	0.68	0.49	9.93	46.85	48.48	11.60	49.03	25.93	92.39	133.90
Ca–SO$_4$	Maximum	8.32	2.67	9.54	161	360	48	150.87	200.17	960	259.25
	Minimum	7.03	0.83	0.99	32.2	80	14.4	71	23.02	134.4	109.8
	Mean	7.72	1.46	3.47	101.25	152	31.44	97.62	76.83	350.4	192.65
	SD	0.53	0.68	3.10	47.93	107.11	12.30	28.06	63.70	304.69	54.16
Ca–Cl	Maximum	8.46	3.5	177.65	149.22	320	87.6	497	252.42	528	323.3
	Minimum	7.23	0.60	0.88	39.1	54	15.6	95.85	17.27	9.6	128.1
	Mean	7.73	1.69	34.80	106.17	154.5	41.1	250.86	116.47	168.8	239.42
	SD	0.56	0.99	70.29	43.42	91.05	27.06	157.76	101.62	202.99	74.15
Mg–HCO$_3$	Maximum	8.11	1.68	4.77	92	110	91.2	142	50.48	187.2	518.5
	Minimum	7.7	0.78	1.88	39.928	62	39.6	79.87	10.18	33.6	237.9
	Mean	7.86	1.14	3.03	66.39	78.66	58.4	109.45	23.91	116.8	335.5
	SD	0.21	0.47	1.53	26.04	27.15	28.50	31.16	23.01	77.59	158.6
Mg–SO$_4$	Maximum	8.06	3.95	18.87	399.69	180	216	372.7	60.67	816	533.75
	Minimum	7.2	1.49	3.66	99.82	76	68.4	106.5	22.14	288	317.2
	Mean	7.73	2.54	9.74	220.70	114.5	127.8	232.9	36.64	528	408.7
	SD	0.39	1.02	6.87	127.08	45.29	62.52	109.0	16.80	225.14	97.44
Na–HCO$_3$	Maximum	8.76	1.48	4.66	222.08	108	44.4	173.95	115.5	134.4	628.3
	Minimum	7.1	0.83	0.77	71.8	40	12	53.25	7.52	14.4	189.1
	Mean	8.06	1.09	2.83	124.6	63.4	30.6	109.8	37.64	59.52	345.5
	SD	0.49	0.24	1.38	45.29	19.48	10.84	44.08	30.43	46.27	143.8
Na–Cl	Maximum	8.35	6.33	35.53	839.84	358	144	1,917	87.68	638.4	664.9
	Minimum	7	1.17	1.22	131.1	56	33.6	150.87	14.61	96	176.9
	Mean	7.60	2.69	7.65	317.82	127.1	78.37	499.77	49.76	264	380.60
	SD	0.43	1.34	8.25	181.0	80.20	34.92	422.62	20.32	152.58	153.02
Na–SO$_4$	Maximum	8.5	8.9	155.44	1,451.99	452	222	1,393.3	170.05	1,920	713.7
	Minimum	6.71	0.53	0.77	80.5	44	8.4	71	9.74	96	48.8
	Mean	7.53	3.55	12.51	503.06	162.05	98.19	401.32	71.23	952.8	344.57
	SD	0.46	2.07	33.85	356.91	95.37	55.75	310.20	47.77	554.6	174.34

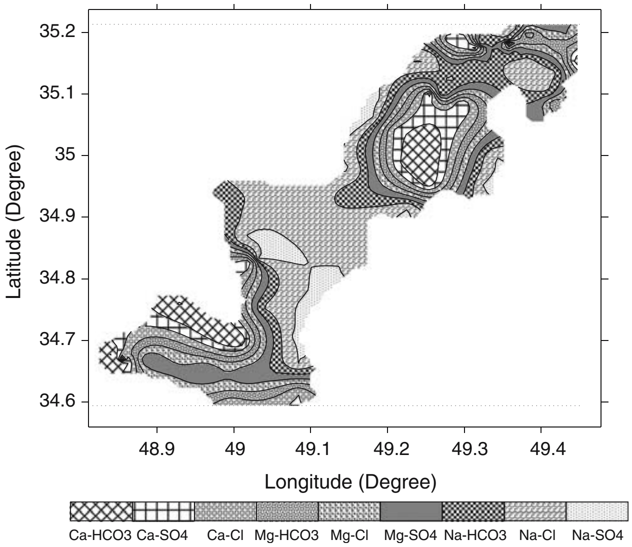


Fig. 5 Spatial distribution of water types in the groundwater of the study area

The second group (group B) comprises some parts of the studied area (Fig. 6). Group B groundwaters show relatively higher TDS contents, with values varying from 1,000 to 5,696 mg l^{-1}. The average value being 2,066 mg l^{-1}. The samples show an increase in Na^+/Cl^- ratio with decreasing Cl^- concentration (Table 3). The Ca^{2+}/Mg^{2+} ratio (average value 1.64) is almost the same as group A (1.60) (Table 3). Sodium is the dominant cation species, representing on average 47% of all the cations. Calcium and Mg^{2+} are secondary in importance representing on average 27 and 24% of all cations. The concentration of SO_4^{2-} for this group of groundwaters, representing on average 36.3% of all the anions. Chloride, HCO_3^- and NO_3^- species are 35.4, 22.5 and 4% respectively. Therefore, groundwater in this group is Na–SO$_4$ and Na–Cl facies. The $Ca^{2+}/(Na^+ + K^+)$ and K^+/Na^+ ionic ratios are very low (Fig. 7) indicating the increasing importance of Na^+ species.

Table 3 Range and mean chemical compositions and standard deviation in two water groups of groundwater samples

		Na/Cl	K/Cl	Ca/Cl	SO_4/Cl	NO3/Cl	HCO_3/Cl	Na/Ca	Na/K	Na/Mg	Ca/Mg	TDS	ESP	SAR	adj SAR
Group A	Mean	1.38	0.03	1.71	0.87	0.35	1.87	2.37	83.80	3.14	1.59	653	4.66	2.35	5.51
	Max	3.6	0.55	6.25	3.50	3.08	8.75	9.65	377.80	10.99	4.50	982	29.78	10.95	28.6
	Min	0.19	0.00	0.24	0.04	0.01	0.35	0.12	0.35	0.35	0.54	250	0.10	0.17	0.5
	SD	0.74	0.08	1.11	0.83	0.49	1.43	2.13	73.08	2.33	0.81	195	5.83	1.99	5.04
Group B	Mean	1.48	0.04	1.02	1.42	0.14	0.72	5.20	161.56	5.82	1.64	2,066	14.25	5.64	15.27
	Max	3.38	0.69	6.54	7.27	0.75	2.12	25.26	926.27	40.68	9.37	5,696	47.22	16.93	54.1
	Min	0.42	0.00	0.20	0.03	0.007	0.06	0.75	1.32	1.05	0.21	1,000	2.34	1.56	4.5
	SD	0.68	0.11	1.05	1.35	0.14	0.41	4.98	203.28	7.43	1.87	1,019	10.84	3.71	10.72

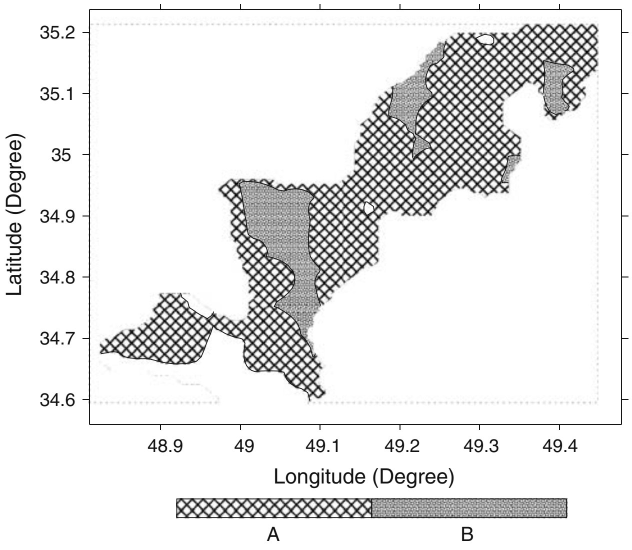


Fig. 6 Spatial distribution of water groups in the groundwater of the study area

Sources of major ions

The concentrations of dissolved ions in groundwater samples are, generally governed by lithology, velocity and quantity of groundwater flow, nature of geochemical reactions, solubility of salts and human activities (Karanth 1997; Bhatt and Saklani 1996). The concentration of carbonates is caused by the CO_2 present in the soil zone formed from the weathering of parent materials. The HCO_3^- may derive from carbonate and silicate minerals dissolution. The plot of $Ca^{2+} + Mg^{2+}$ vs HCO_3^- in Fig. 8a shows that about half of the data falls below the equiline (1:1). It suggests that an excess of alkalinity of the waters has been balanced by alkalies ($Na^+ + K^+$). The exceed of alkaline earth elements ($Ca^{2+} + Mg^{2+}$) over HCO_3^- in some samples reflects an extra source of Ca^{2+} and Mg^{2+} ions. It might have been balanced by Cl^- and SO_4^{2-} and/or supplied by silicate weathering (Zhang et al. 1995). Further ($Ca^{2+} + Mg^{2+}$) versus total cations shows that the data lie far below the theoretical line depicting an increasing contribution of alkalies to the major ions (Fig. 8b). The increase in alkalies corresponds to a simultaneous increase in $Cl^- + SO_4^{2-}$ (Fig. 8c) indicating a common source for these ions. The excess of Na^+ over K^+ (Table 3) (Fig. 8d) is because of the greater resistance of K^+ to weathering and its fixation in clay minerals. The dominance of Na^+ suggests that the ions result from silicate weathering and/or dissolution of soil salts. The excess of Na^+ over Cl^- in both water types (Table 3) indicates silicate weathering (Stallard and Edmond 1983), and also suggested that the higher concentration of alkalize is from sources other than precipitation (Singh and Hasnain 1999). Groundwaters in the area have a higher average ratio (0.57) of ($Na^+ + K^+$)

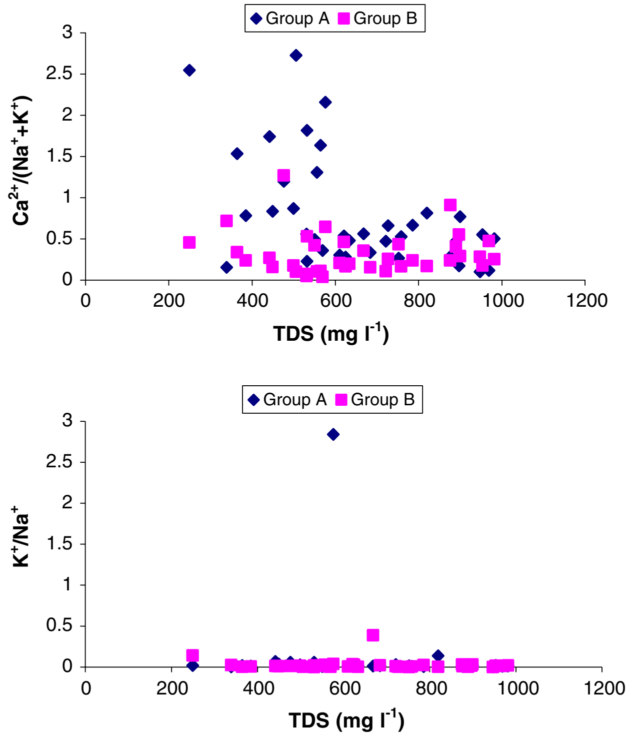
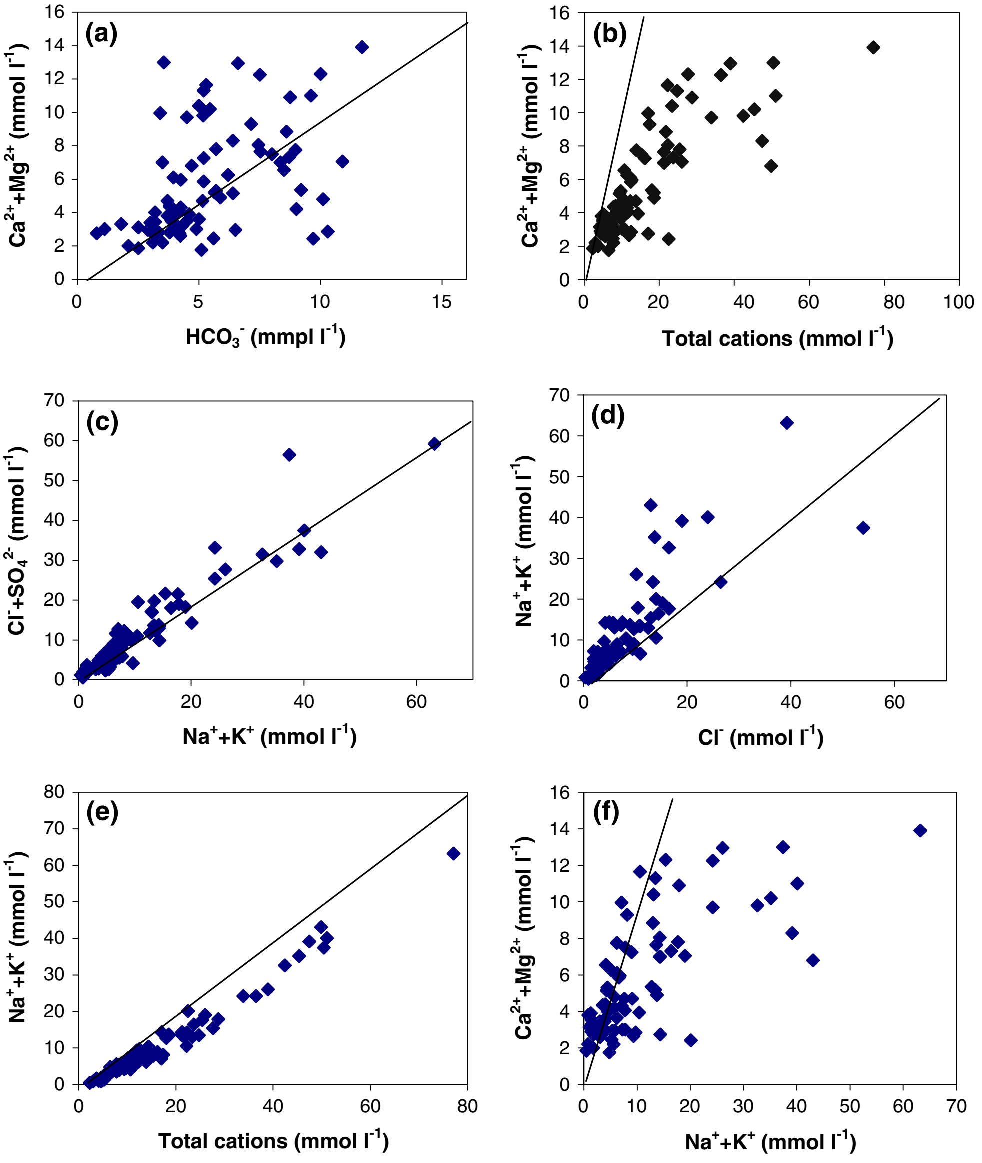


Fig. 7 Relationship between cation ratios and total dissolved solids for Tajarak groundwaters

Fig. 8 Scatter diagrams



versus total cations (Fig. 8e) depicting the contribution of cations via silicate weathering and/or soils (Datta and Tyagi 1996). Molar Na^+/Ca^{2+} ratios (Table 3) for both groups are more than unity, indicating a deficiency of Ca^{2+}. This may be caused by the precipitation of $CaCO_3$ and/or ion exchange process. The evaporation also results in the precipitation of $CaCO_3$. Another important factor for the deficiency of Ca^{2+} in the groundwaters is the ion exchange process, which generally indicates the excess of alkalize over alkaline earth elements (Fig. 8f)

The functional sources of dissolved ions can be assessed by plotting the samples according to the Gibbs plot. Gibbs (1970) suggested that a simple plot of TDS versus the weight ratio of $Na^+/(Na^+ + Ca^{2+})$ could provide information on the relative importance of the major natural mechanisms controlling groundwater chemistry: (1) atmospheric precipitation, (2) rock weathering and (3) evaporation and precipitation. A Gibbs plot (Fig. 9) of data indicates that groundwater chemistry in group A may have acquired their chemistry mainly from rock weathered materials derived from the underlying rocks. Also, evaporation and precipitation are dominant factors that control the groundwater chemistry in group B water samples. Evaporation greatly increases the concentrations of ions formed by chemical weathering, leading to higher salinity. As the anthropogenic activities also influence the role of evaporation, leading to an increase in Na^+ and Cl^- and thus TSD (Hem 1991), the samples fall in an environment that tends towards a semi-arid climate. Such an environmental condition is also

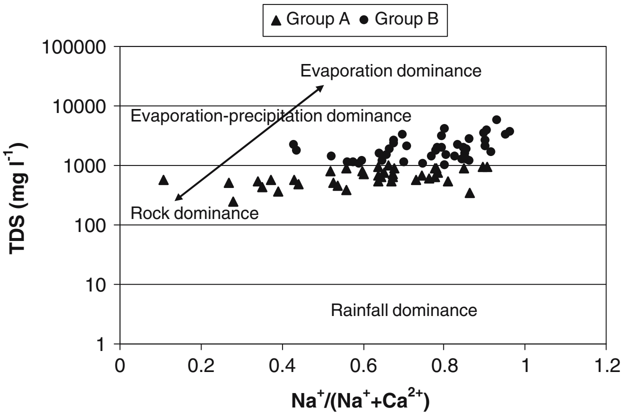


Fig. 9 Gibbs plot indicating mechanisms governing groundwater chemistry

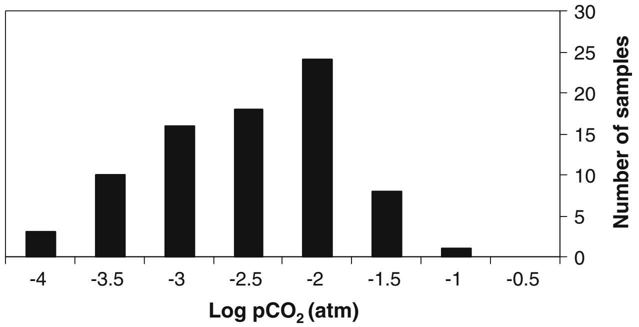


Fig. 10 Distribution of equilibrium partial pressure of CO$_2$ of the groudwater in the study area

responsible for moving the chemistry of groundwaters of the study area towards the zone of evaporation dominance.

Hydrochemical processes

Water–rock interaction

The different compositions of groundwater types result from several hydrochemical processes. Interactions between groundwater and host rocks are the main processes controlling chemical characteristics of groundwater in the studied area. These processes generally include chemical weathering of minerals, dissolution–precipitation of secondary carbonates and ion exchange between water and clay minerals (Njitchoua et al. 1997). The first group, with high HCO$_3^-$ concentration is probably formed as a result carbonate weathering. The slight supersaturation of groundwater with respect to dolomite and calcite is probably caused by the combined effect of dissolution of these minerals and the evapotranspiration, both processes causing an increase in the concentrations of Ca^{2+}, Mg^{2+} and HCO$_3^-$. The composition of group B is due to evapotranspiration of infiltrating rainwater and the dissolution of calcite, dolomite and gypsum.

Figure 10 shows the distribution of the computed pCO$_2$ values for groundwater samples. The values range from 2.34×10^{-4} to 1.07×10^{-1} with a mean value of 1.41×10^{-2} (atm), which is above the pCO$_2$ of the earth's atmosphere ($10^{-3.5}$), due to the decay of organic matter and root respiration in the soil zone. This suggests that the groundwater system is open to soil CO$_2$ (Njitchoua et al. 1997). In a closed system, pCO$_2$ ranges between $10^{-2.5}$ and $10^{-6.4}$ atm (Appello and Postma 1994; Tellam 1994; Stigter et al. 1998). As dissolved CO$_2$ gas pressure of the waters is higher than that of the atmosphere, the waters are supersaturated with respect to carbonate minerals.

It was found that the groundwater is undersaturated with respect to gypsum, anhydrite and halite and oversaturated with respect to calcite, aragonite and dolomite.

Figure 11 shows the plots of SI against TDS for all the investigated waters. Groups A and B differ from each other in their behaviour with respect to carbonate minerals. The entire group A (expect two samples) groundwaters are supersaturated with respect with calcite and dolomite (Fig. 11), suggesting that these carbonate mineral phases are present in the corresponding host rock. In contrast, all the group B groundwater is less saturated with respect to calcite and dolomite, indicating that these carbonate mineral phases may have influenced the chemical composition of this group of groundwaters. Both groups are under saturated with respect to gypsum, halite (Table 2). Compared with group A groundwaters, the geochemistry of group B may be more complex, owing to the occurrence within the corresponding host rock of mineral phases other than primary aluminosilicatess, such as, calcite, and dolomite. Subsequent reactions involving these mineral phases such as dissolution–precipitation of secondary carbonate minerals, and cation exchange between groundwater and clay minerals may have influenced the observed chemical composition of these waters.

Cation exchange

The cation exchange between Ca^{2+} or Mg^{2+} and Na$^+$ may also explain the excess Na$^+$ concentration (Stimson et al. 2001). Those samples with a value of Na$^+$/Cl$^-$ ratio higher than one also show a deficit in Ca^{2+} + Mg^{2+} and this is consistent with a Ca^{2+}–Na$^+$ cation

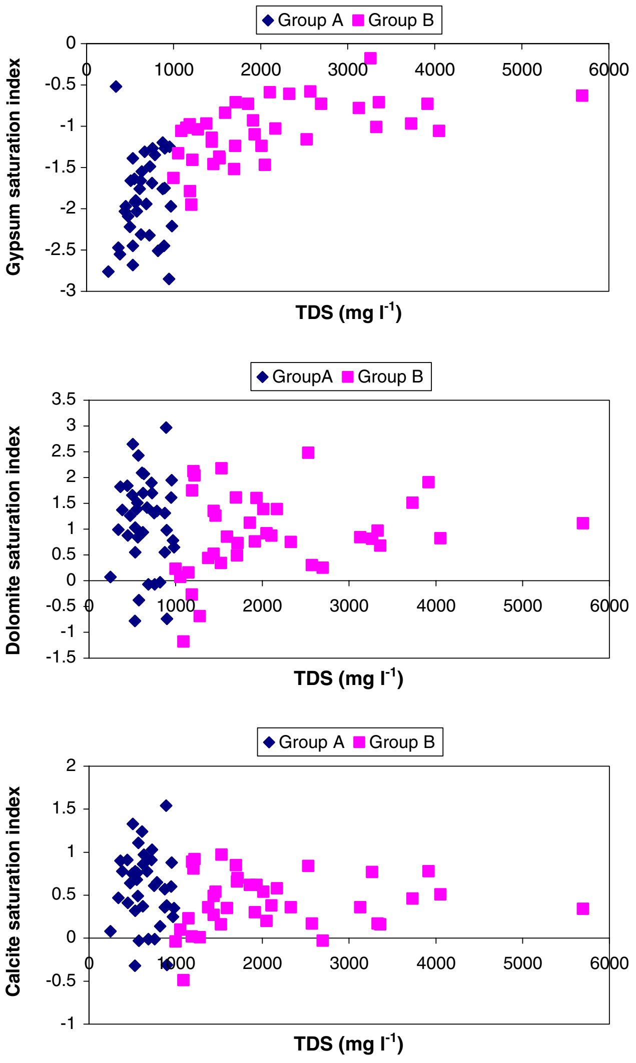


Fig. 11 Plots of saturation indices with TDS in groundwater in the study area. Group A low salinity water and group B high salinity groundwaters

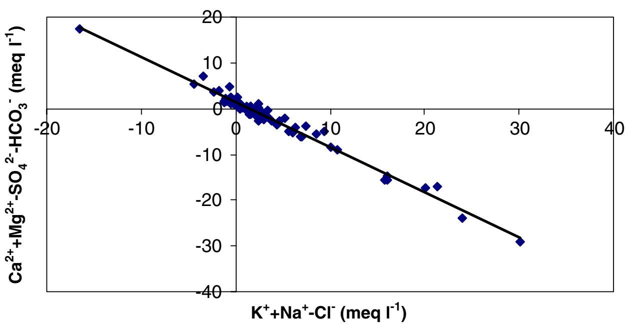


Fig. 12 Plot of $Ca^{2+} + Mg^{2+} – SO_4^{2-} – HCO_3^-$ and $Na^+ + K^+ – Cl^-$ in the groundwater of the study area

exchange process which leads to a softening of the water (Hidalgo et al. 1995. Hidalgo and Cruz-Sanjulian 2001). A Na–HCO_3 water type is usually an indication of cation exchange processes (Tijani 2004). Such water type is referred to as exchange waters which imply that there is more HCO_3^- than the available alkaline–earth cations (Ca^{2+} + Mg^{2+}) in equivalent concentrations (Tijani 2004). These excess bicarbonate ions then cause a release of the alkali ions (usually Na^+) into the solution by the exchange reaction with the exchange sites.

To test the possibility that cation exchange significantly affects groundwater compositions, plots of $Ca^{2+} + Mg^{2+} – HCO_3^- – SO_4^{2-}$ in the function of $Na^+ + K^+ – Cl^-$ were examined (Fig. 12). $Na^+ + K^+ – Cl^-$ represents the amount of $Na^+ + K^+$ gained or lost relative to that provided by chloride salts dissolution (mostly halite dissolution), while $Ca^{2+} + Mg^{2+} – SO_4^{2-} – HCO_3^-$ represents the amount of Ca^{2+} and Mg^{2+} gained or lost relative to that provided by gypsum, calcite and dolomite dissolution. In the absence of these reactions, all data should plot close to the origin (McLean et al. 2000). If these processes are significant composition controlling process, the relation between these two parameters should be linear with a slope of –1. Figure 12 indicates an increase in $Na^+ + K^+$ related to a decrease in $Ca^{2+} + Mg^{2+}$ or an increase in $HCO_3^- + SO_4^{2-}$. All data plot, as expected, close to a straight line ($r = 0.988$) with a slope of –0.980, clearly points to the existence of cation exchange (Garcia et al. 2001; Jalali 2005b). Thus, ion exchange is also responsible for the increase of Na^+ in both water groups.

Enrichment of groundwater in Ca^{2+} as a result of both Na^+ –Ca^{2+} exchange and dissolution of calcite in the unsaturated zone (caused by nitrification and increase in CO_2 concentration) created favourable conditions for calcite precipitation. Cation exchange contributes partly to the groundwater enrichment in Ca^{2+}. The remaining Ca^{2+} excess must be attributed to the dissolution of calcite due to anthropogenic pollution (oxidation of ammonium with increasing water acidity, and increased CO_2 concentration).

According to Stumm and Morgan (1996), nitrification (microbial oxidation of ammonium) in the unsaturated zone, ammonium is transformed into NO_3^- as described by the following equation,

$$NH_4^+ + 2O_2 = NO_3^- + 2H^+ + H_2O.$$

Correlation between NO_3^- and Ca^{2+} concentrations in groundwater ($r = 0.34$) provides evidence for development of this process. Increased acidity during nitrification leads to carbonate dissolution, with resulting Ca^{2+} enrichment in groundwater. Greater mineralization is generally associated with higher NO_3^- concentrations in the groundwaters.

Anthropogenic inputs

The common groundwater contaminant in the area was NO_3^-. Nitrate is the most common form of nitrogen that occurs in surface and groundwater. Because of its anionic form, NO_3^- is very soluble and mobile in aqueous solution. It represents the oxidized end product in the nitrogen cycle of atmosphere, vegetation, upper soil and soil water zones. Concentrations of NO_3^- are the result of different pollution processes, sewage effluents, industrial wastes and agricultural flows (Hern and Feltz 1998; Hao and Change 2002). Part of the nitrate contamination is from natural nitrogen in the soil, released to the unsaturated zone due to intense cultivation of virgin soil. Other sources of NO_3^- can be derived from fertilizers, wastewater irrigation and breakdown of remnants of the growing crops (Kanfi et al. 1983; Ronen et al. 1983). The lines of equal NO_3^- concentrations are shown in Fig. 3c. The results indicate that in 31% of samples the concentration of NO_3^- is above the recommended guidelines of the World Health Organization (50 mg $l^{-1}NO_3^-$). The Ca–Cl, Ca–SO_4 and Na–SO_4 water types are associated with high NO_3^- pollution (Table 2). The high concentration of NO_3^- in the studied area could be related to wastewater leakage from industrial activities, urbanization and agricultural practices (Jalali 2005a). The use of NO_3^- fertilizers in the intensively cultivated area of the study area contributes to the increase of the concentration of NO_3^-.

Salinization

One of the problems concerning groundwater quality is rapid salinization of water resources. Anthropogenic contamination is a major cause of salinization and water quality degradation. Salinity is the total amount of inorganic solid material dissolved in any natural water and water salinization refers to an increase in TDS and overall chemical content of water (Richter and Kreitler 1993). The high salinity in irrigation water reduces crop yield and irrigation water must be applied in excess quantities to leach salts from the soil. Changes in groundwater salinity and overall chemical composition occur along flow paths from recharge to discharge area due to either natural and/or anthropogenic causes (Richter and Kreitler 1993). Water that enters the soil is subject to chemical, physical and biological changes. Several hydrochemical processes including movement of groundwater through rocks containing soluble mineral materials, concentration of water by evaporation and industrial and municipal waste disposal may cause an increase in dissolved solids. Moreover, irrigation with wastewater (in some parts of the studied area), which is more saline than regional groundwater increases the rate of salinization of groundwater. Groundwater classifications based on TDS (Freeze and Cherry 1979) indicates that 52% of the samples are brackish water (1,000–10,000 mg l^{-1}) with TDS ranging from 1,000 to 5,696 mg l^{-1}. Most of the fields are flood irrigated, with poorly managed irrigated systems, resulting in low irrigation efficiency. The fields are irrigated mainly during the late spring and summer months and the overall amount of water applied to the field estimated at 1,000 mm $year^{-1}$. Within the study area, the salt loading resulting from the application of groundwater for irrigation having 250–5,696 mg l^{-1} of TDS would be 2.5–57 Mg ha^{-1} $year^{-1}$. The relatively high salinity may be due to the lithologic composition of these localities, in addition, the leaching and dissolution of the soil salts and chemical fertilizers by irrigation waters.

Quality of irrigation water

The water quality evaluation in the area of study is carried out to determine their suitability for agricultural purposes. Sodicity, salinity and toxicity were used for evaluating water quality for irrigation purposes (Shainberg and Oster 1976). The Food and Agricultural Organization (FAO) guidelines were used to interpret water quality for irrigation (Ayers and Westcot 1985). Parameters such as EC, sodium adsorption ratio (SAR), adjusted SAR (adj SAR) and the exchangeable sodium percentage (ESP) were used to assess the suitability of water for irrigation purposes.

Sodicities

Sodium concentration is important when evaluating the suitability of groundwater for irrigation. High concentrations of Na^+ are undesirable in water because Na^+ adsorb onto the soil cation exchange sites, causing soil aggregates to disperse, reducing its permeability (Tijani 1994). Stigter et al. 1998 demonstrated that irrigation with Na-enriched water results in ion exchange reactions: uptake of Na^+ and release of Ca^{2+} and Mg^{2+}. In contrast, irrigation with Ca-enriched water releases Na^+ that is bound to the adsorption sites on clay minerals. The SAR which indicates the effect of relative cation concentration on sodium accumulation in the soil was calculated as follows

$$\mathrm{SAR} = \frac{\mathrm{Na}^+}{\sqrt{\frac{\mathrm{Ca}^{2+}+\mathrm{Mg}^{2+}}{2}}}.$$

The ionic symbols indicate concentrations of the ions in the water in meq l^{-1}. Calculation of the SAR for given water provides a useful index of the Na^+ hazard of the water for soils and crops. The calculated SAR ranged from 0.17 to 11 and 1.6 to 5.6 for group A and B, respectively (Table 3). The FAO guidelines (Ayers and Westcot 1985) refers to the sodicity problem as permeability. Excessive Na^+ causes soil mineral particles to disperse and water penetration to decrease. High Na^+ concentrations become a problem when infiltration rate is reduced to the extent the crop is not adequately supplied with water or when the hydraulic conductivity of the soil profile is too low to provide adequate drainage. The problem of Na^+ is most severe with montmorillonitic soils and least with kaolinitic. Another property related to the Na^+ hazard of irrigation waters is the HCO_3^- concentration. Bicarbonate toxicities associated with some waters generally arise from deficiencies of iron or other micronutrients caused by the resultant high pH (Bohn et al. 1985). Precipitation of Ca^{2+} carbonate from such waters lowers the concentration of dissolved Ca^{2+}, increases the SAR and increases the exchangeable Na^+ level of the soil (Bohn et al. 1985). The adj SAR is a value corrected to account for the removal of Ca^{2+} and Mg^{2+} by their precipitation with HCO_3^- and CO_3^{2-} ions in the water added and adj SAR is defined as

$$\mathrm{adjSAR} = \frac{\mathrm{Na}^+}{\sqrt{\frac{\mathrm{Ca}^{2+}+\mathrm{Mg}^{2+}}{2}}}[1+(8.4-\mathrm{pH}_c)]$$

$$\mathrm{pH}_c = (pk_2 - pk_c) + p(\mathrm{HCO}_3^- + \mathrm{CO}_3^{2-}) + p(\mathrm{Ca}^{2+} + \mathrm{Mg}^{2+})$$

where p refers to the negative logarithm, k_2 is the second dissociation equilibrium constant of the carbonic acid, k_c is the solubility equilibrium constant for calcite. Concentration of Ca^{2+}, Mg^{2+}, HCO_3^- and CO_3^{2-} in meq l^{-1} (Ayers and Westcot 1985). The calculation for adj SAR produced higher values than for the SAR (Table 3). This indicates that concentration of HCO_3^- in water samples may be high. Bicarbonate causes precipitation of Ca^{2+} as calcium carbonate, resulting in a higher SAR in the water (high Na^+ hazard), because of the lowered Ca^{2+} content. The HCO_3^- ion in soil solution harms the mineral nutrition of the plant through its effect on the uptake and metabolism of nutrients.

For evaluating water quality for irrigation, in terms of Na^+ hazard, adj SAR, EC and clay mineralogy are considered (Ayers and Westcot 1985). In group B of water samples, with respect to the mineralogy of soils that are dominated with illite and vermiculite (Jalali 2006a), in 24% of samples, adj SAR are <8, in 40% of samples, adj SAR is 8–16 and in 36% of samples is larger than 16. Therefore, some of the water samples have no problem and in 40 and 37% of the water sample the intensity of problem are moderate to severe, respectively. Only 10% of water samples in group A the adj SAR is more than 8, indicating that most of the water in this group have no problem.

The ESP of soils can be predicted quite well from the following equation

$$\mathrm{ESP} = \frac{100(-0.0126 + 0.01475\mathrm{SAR})}{1 + (-0.0126 + 0.01475\mathrm{SAR})}.$$

The expected ESP for soils that are irrigated with groundwater would be in range of 1–15 and 1–24 for group A and B, respectively (Table 3). The ESP indicates the potential decline in soil structure. High concentrations of Na^+ are undesirable in water because Na^+ adsorb onto the soil cation exchange sites, causing soil aggregates to disperse, reducing its permeability (Tijani 1994).

Ca^{2+}/Mg^{2+} ratio

The ratio of Ca^{2+}/Mg^{2+} in the soil–water may be used to predict a potential Ca^{2+} deficiency (Ayers and Westcot 1985). In an Mg^{2+} dominated water (ratio of $Ca/Mg < 1$) or an Mg^{2+} soil (soil–water ratio of $Ca^{2+}/Mg^{2+} < 1$), the potential effect of Na^+ may be slightly increased. In other words, a given SAR value will show slightly more damage if the Ca^{2+}/Mg^{2+} ratio is less than 1. The lower the ratio, the more damaging is the SAR. Results of this study showed that 35% of water samples have Ca^{2+}/Mg^{2+} ratio less than 1. The Ca^{2+}/Mg^{2+} ratio in group B is slightly higher than group A (Table 3).

Salinity

The conductivity values ranged from 0.39 to 8.9 dS m^{-1}. Water quality in discharge zone does not meet the international standards for irrigation water quality. The EC should not exceed 3.0 dS m^{-1}. Normally, irrigation water with an EC of < 0.7 dS m^{-1} poses little or no threat to most crops while EC > 3.0 dS m^{-1} may restrict their growth (Tijani 1996). Most annual crops are sensitive to high concentrations of $Cl^-$and Na^+. The salinity hazard for some water wells is classified as medium (71.5%), high (22.5%) and only 6% of water samples had low salinity contamination. Figure 13 shows spatial distribution of EC in groundwaters. As it can be seen samples from south and west part of the studied areas have low salinity hazard while the samples from north east of the studied areas had very high salinity and are unsuitable for irrigation. The relatively high salinity may be due to the lithologic composition of these localities, in addition, the leaching and dissolution of the soil salts and chemical fertilizers by irrigation waters. This irrigation water quality also leads to high leaching requirements. The leaching requirement is the amount of additional irrigation water required to avoid a harmful build-up of salts in the root zone of plants (Luedeling et al. 2005).

Water classification

In order to identify the availability of waters for irrigation use, the Wilcox classification diagram (1955) has been used (Fig. 14). This graph is based on the EC and SAR. According to this graph water classes in most of water samples in group A are C3-S1 and C2-S1, while in group B C3-S1, C4-S1 and C4-S2 are dominate water classes. Salinity in C3-S1 classes is high. Water cannot be used on soil with restricted drainage. Even with adequate drainage, special management for salinity control may be required, and plants with good salt tolerance should be selected. Sodium in this water class is low. Water can be used for irrigation on almost all soils with little danger of the development of Na^+. In C2-S1 classes, salinity is medium. Water can be used if a moderate amount of leaching occurs. Plants with moderate salt tolerance can be grown. In C4-S2 classes, salinity is very high. Water is not suitable for irrigation under ordinary conditions but may be used under very special circumstances. The soil must be permeable. Sodium in this water class is medium. Water may be present a moderate Na^+ problem in fine-textured (clay) soils. This water can be used on coarse-textured (sandy) soils. Therefore, the salinity hazard in groundwater samples is regarded as high to medium, but the Na^+ hazard is regarded as low to medium.

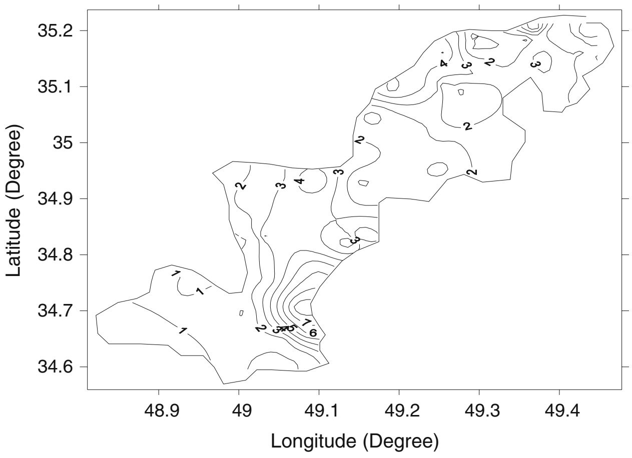


Fig. 13 Spatial distribution of EC (dS m^{-1}) in groundwater of the study area

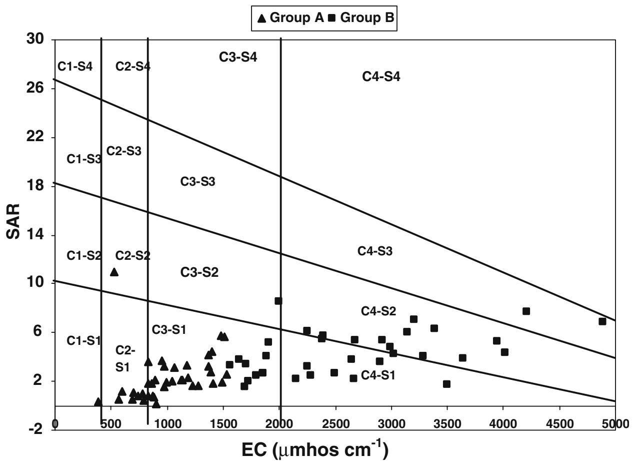


Fig. 14 Diagram of sodium adsorption ratio and salinity for the calssification of groundwater for irrigation purposes

Environmental impact of irrigation with poor-quality water in arid and semi-arid regions

A characteristic of arid and semi-arid regions is low rainfall and the necessity of irrigation. The shortage of quality water resources is becoming an important issue in arid and semi-arid regions of the world. In these regions, the availability of non-saline river or canal water is limited and prioritized to supplying urban areas (Beltran 1999). Poor-quality groundwaters are a common feature of arid and semi-arid regions. Groundwater is commonly the only source of irrigation, although its quality is usually low because of limited rainfall and high rates of evaporation. Thus, there is an increasing need to irrigate using low- to medium-quality groundwater.

Poor-quality water may lead to leaching of nutrients from soil. Leaching of nutrients is of major environmental concern, as high concentration of some ions in the drinking water is harmful to human health. When combined irrigation and rainfall exceeds the crop water requirement, the excess soil water drains downward, carrying with it soluble salts including K^+. The degree of K^+ leaching is dependant on the Ca^{2+} concentration of the leaching waters: high-quality waters generate little displacement of K^+ (Kolahchi and Jalali 2006). The presence of Ca^{2+} in irrigation water and soil minerals able to release Ca^{2+} is important in determining the amount of K^+ leached from soils.

The ionic composition of such water affects cation exchange in the soil complex. Irrigation with water in which the concentrations of Ca^{2+}, Mg^{2+} and Na^+ are higher than those in high-quality water leads to an increase in K^+ desorption and leaching. Bar-Tal et al. (1991) showed that irrigation water with high salinity can leach native and applied K^+ from the soil. Jalali and Rowell (2003) reported losses equivalent of 63–170 kg K^+ ha^{-1} (12–33% of the initial exchangeable K^+) when 196 mm of solution containing 2–30 meq l^{-1} $CaCl_2$ were applied to soil column in the laboratory. Kolahchi and Jalali (2006) studied the effect of water quality on the leaching of K^+ from sandy soil. They reported that large amounts of K^+ are leached from soils in areas where crops are irrigated with water that contains significant concentrations of Ca^{2+} and other cations.

Therefore, an increase in K^+ concentration can be expected in groundwater within infiltration areas subjected to agricultural land use. Such increases can even lead to a breach of the drinking water limit for potassium (12 mg l^{-1}) (Griffioen 2001; WHO 1993).

Low irrigation efficiency contributes large additions of water to the hydrological system. The recharge coefficient of the irrigation water in the studied area was estimated from the salt concentration factor and on average, a factor of about 1.41 was found, indicating that 29% of the irrigation water is lost by evapotranspiration and the rest 71% infiltrates towards the groundwater (Jalali 2006b). A soil at its wilting point at the end of November requires rain to bring it back to field capacity, and any extra water will then leach through. For a sandy loam with a wilting point of 0.05 cm^3 H_2O cm^{-3} soil and a field capacity of 0.2 cm^3 cm^{-3} (Rowell 1994), the top 60 cm of soil needs 90 mm of water to bring it back to field capacity. Thus, of the 1,000 mm, about 620 mm moves through the 0–60 cm layer during irrigation (Jalali 2006b). If Na^+ or NO_3 concentration are maintained at 35 mg l^{-1} (Merikhpour and Jalali 2005) and 18 mg l^{-1} (Jalali 2005c), respectively, the amount of Na^+ and NO_3^- leached to below 60 cm is 217 and 52.2 kg ha^{-1} $year^{-1}$, respectively. This is higher than the value found in leaching of Na^+ or NO_3^- by rainfall (Jalali 2006b), indicating the importance of irrigation water in increasing the concentration of ions in groundwater in discharge area. Stites and Kraft (2001) calculated NO_3^- and Cl^- loading to groundwater from an irrigated north-central US sand–plain vegetable field. They found that loading to groundwater was 165 kg ha^{-1} NO_3–N and 111 kg ha^{-1} Cl^- for sweet corn (*Zea mays* L.) and 228 kg ha^{-1} NO_3–N and 366 kg ha^{-1} Cl^- for potato (Solanum tubersum).

Arid and semi-arid conditions create irrigation-return flow to become concentrated in some chemical constituents because of a number of processes such as evapotranspiration, mineral dissolution, fertilizer application and agricultural residues (Cardona et al. 2004). The net loss in Ca^{2+} and HCO_3^- are common as a result of precipitation accompanied by a net gain of Na^+, K^+, Cl^-, SO_4^{2-} and NO_3^- (Richter and Kreitler 1993).

Therefore, irrigation with poor-quality water requires a comprehensive analysis even beyond the area where water is applied. Beltran (1999) suggested that the sustainable use of poor-quality water in irrigated agriculture of arid and semi-arid regions requires the control of soil salinity at the field level, a decrease in the amount of drainage water, and the disposal of the irrigation return flows in such a way that minimizes the side effects on the quality of downstream water resources.

Conclusion

The weathering of minerals is of prime importance in controlling the groundwater chemistry. The chemical analyses of some groundwater samples from Tajarak aquifer indicate that at least two types of groundwaters are present within the study area. The first group is chemically of Ca–HCO_3 and Na–HCO_3, with low TDS. The groundwaters from this group are recharged through direct infiltration of rain. The groundwater evolution is mainly the result of weathering of minerals. The second group consists of relatively high salinity groundwaters of Na–SO_4 and Na–Cl water types. The evolution of these waters may be controlled by more complex processes involving: evaporation, precipitation and dissolution of carbonate minerals, cation exchange reactions between groundwater and clay minerals. In arid and semi-arid regions, agriculture is mainly limited by the availability of suitable irrigation water. Sustainable productivity of the soils in these regions depends mainly on the maintenance of the soil's salt concentration at a level which is not limiting the growth of plants. Excessive Na^+ causes soil mineral particles to disperse and water penetration to decrease. High Na^+ concentrations become a problem when infiltration rate is reduced to the extent the crop is not adequately supplied with water or when the hydraulic conductivity of the soil profile is too low to provide adequate drainage. The problem of Na^+ is most severe with montmorillonitic soils and least with kaolinitic.

Groundwater proved to be of rather poor quality, though it is used for irrigation due to limited fresh water resources. The most dominant classes C3-S1, C4-S1 and C4-S2 were found in the studied area. The salinity hazard for water wells is classified as medium, high and some show very high salinity. The Na^+ hazards in some samples are low, an indication that these waters are suitable for irrigation in almost all soils with little danger of the development of Na^+ problem. Therefore, salinity is the principal concern in irrigated agriculture in Tajarak areas. Special management for salinity control is required and plants with good salt tolerance should be selected. Maintaining soil organic matter levels and appropriate leaching is necessary to avoid the build-up of salts over time.

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