Assessment of groundwater quality and health risk in drinking water basin using GIS
Şehnaz Şener, Erhan Şener and Ayşen Davraz

ABSTRACT
Eğirdir Lake basin was selected as the study area because the lake is the second largest freshwater lake in Turkey and groundwater in the basin is used as drinking water. In the present study, 29 groundwater samples were collected and analyzed for physico-chemical parameters to determine the hydrochemical characteristics, groundwater quality, and human health risk in the study area. The dominant ions are $\text{Ca}^{2+}$, $\text{Mg}^{2+}$, $\text{HCO}_3^{-}$, and $\text{SO}_4^{2-}$. According to Gibbs plot, the predominant samples fall in the rock–water interaction field. A groundwater quality index ($\text{WQI}$) reveals that the majority of the samples falls under good to excellent category of water, suggesting that the groundwater is suitable for drinking and other domestic uses. The Ca-Mg-HCO$_3$, Ca-HCO$_3$, Ca-SO$_4$-HCO$_3$, and Ca-Mg-HCO$_3$-SO$_4$ water types are the dominant water types depending on the water–rock interaction in the investigation area. Risk of metals to human health was then evaluated using hazard quotients ($\text{HQ}$) by ingestion and dermal pathways for adults and children. It was indicated that $\text{As}$ with $\text{HQ}_{\text{ingestion}} > 1$ was the most important pollutant leading to non-carcinogenic concerns. It can be concluded that the highest contributors to chronic risks were $\text{As}$ and $\text{Cr}$ for both adults and children.

Key words | drinking water quality, Eğirdir Lake basin, health risk assessment, WQI

INTRODUCTION
Groundwater and surface waters are the major sources of water around the world. The economic development and ecological stability of many countries heavily depends on clean and adequate water. Groundwater is a valuable natural resource and is believed to be comparatively much cleaner and free from pollution than surface water (Armon & Kott 1994; Arya et al. 2012; Boateng et al. 2016). Also, due to a lack of surface water, groundwater has played a major role in meeting drinking and irrigation demands in arid and semi-arid regions (Khorsavi et al. 2016). The need for groundwater is greater than ever before, related to industrial and agricultural developments (Kaviarasan et al. 2016). Groundwater quality is an important factor for drinking, domestic, industrial, and agricultural usages (Babiker et al. 2007). The quality of groundwater is affected by natural sources or numerous types of human activity (Kavaf & Nalbantçıl 2007; Nas & Berktay 2010; Varol & Davraz 2015a). Point and non-point pollution sources such as fertilizers, effluent run-off from industries, chemical dumping sites, and domestic sewage cause groundwater to become polluted and to create health problems (Rohul-Amin et al. 2012; Nalbantçılı & Pınarkara 2015). Therefore, systematic monitoring of the water quality parameters controlling hydrochemical processes are essential for sustainable usage of the water.

Hydrochemical studies of groundwater are providing a better understanding of possible changes in quality as development progresses (Varol & Davraz 2015a). A knowledge of hydrochemistry is important to assess the groundwater quality in any area in which groundwater is used for both irrigation and drinking needs (Srinivas et al. 2013). The conventional techniques such as trilinear plots, statistical
techniques are widely accepted methods to determine the quality of water (Kumar et al. 2015). In addition, numerous water quality indices have been formulated all over the world, such as the US National Sanitation Foundation Water Quality Index (NSFWQI) (Brown et al. 1970), Canadian Council of Ministers of the Environment Water Quality Index (CCMEWQI) (Khan et al. 2003), British Columbia Water Quality Index (BCWQI), and Oregon Water Quality Index (OWQI) (Abbbasi 2002; Debels et al. 2005; Kannel et al. 2007) to assess water quality. These indices are one of the most effective ways for water quality information provision to the public, concerned authorities, or policy-makers for water quality management and are considered to be one of the simplest methods used for overall water quality assessment (Shabbir & Ahmad 2015).

Groundwater contamination by inorganic and organic compounds of natural or anthropogenic origin represents a serious global environmental problem since groundwater is used as a significant drinking water source worldwide. The identification and characterization of associated human health risks are important problems that need to be addressed by environmental and medical geochemistry (Rapant & Krêmová 2007). It is known that water pollution may become a significant threat to human health (Davies 1983). Nearly 25,000 people die of such water pollution problems every day and one-third of urban inhabitants in developing countries cannot access safe drinking water (Yin & Deng 2006; Li & Ling 2006). Generally, drinking water containing different anions and heavy metals has significant adverse effects on human health either through deficiency or toxicity due to excessive intake (Varol & Davraz 2015b). For example, arsenic represents one of the most potentially toxic inorganic contaminants of groundwater worldwide (Focazio et al. 1999; Smith et al. 2000; Lin et al. 2002; Ahmed et al. 2004; Bundschuh et al. 2004; Tollestrup et al. 2005; Li et al. 2007; Rapant & Krêmová 2007; Kavcar et al. 2009; Phan et al. 2010). Fluoride, which generally occurs in nature, is beneficial to human health in trace amounts, but can be toxic in excess (Varol & Davraz 2015b). Risk assessment is the methodological approach in which the toxicity of a chemical is identified, characterized, and analyzed. Current knowledge of element and compound toxicity enables a descriptive (qualitative) risk assessment based on the identification of adverse effects (chronic, carcinogenic) and the quantitative assessment of risk level (calculation and map presentation of health risk) (Rapant & Krêmová 2007).

The geographic information system (GIS), a high performance computer-based tool, is playing a critical role in water resource management and pollution study. GIS represents a technological advancement in terms of overlay mapping techniques (Igboekwe & Akankpo 2011). Also, many studies have indicated that GIS is a powerful tool to assess water quality (Butler et al. 2002; Skubon 2005; Asadi et al. 2007; Babiker et al. 2007; Yammani 2007; Rangzan et al. 2008; Jeihouni et al. 2014; Krishnaraj et al. 2015; Varol & Davraz 2015a). GIS applications are used widely in groundwater studies, such as site suitability analyses, managing site inventory data, estimating vulnerability of groundwater to pollution potential from non-point sources of pollution, modeling groundwater movement, modeling solute transport and leaching, and integrating groundwater quality assessment models with spatial data to create spatial decision support systems (Engel & Navulur 1999). Additionally, mapping water quality indices within a GIS framework will be a useful tool for water quality management (Shabbir & Ahmad 2015).

The present study was carried out to evaluate the hydrochemical characteristics, groundwater quality, and human health risk in Eğirdir Lake basin, which is one of the important drinking water basins in Turkey. In the study area, groundwater is used as drinking and irrigation water. There are several point and non-point pollution sources and groundwater quality is under threat in the region. Hence, this study has great importance for the identification of management options for the sustainable usages of the groundwater.

**MATERIALS AND METHODOLOGY**

**Study area**

The Eğirdir Lake catchment area is located within the Lake District in the southwest of Turkey (Figure 1). It is a sub-basin of the Antalya Basin which is one of the 26 major watersheds of Turkey. Eğirdir Lake is an indispensable water source for the region because it is the second largest...
freshwater lake with a 482 km² surface area and available water capacity. The lake has different usages, such as irrigation, tourism, and fishing, and also for supplying the drinking water needs of Isparta city. The main recharging of the lake is rainfall, surface flow from streams, and underground flow from aquifers within the Eğirdir Lake basin.

There are two main groundwater basins, Uluborlu-Senirkent and Yalvaç-Gelendost, with a combined area of 525 km². Yalvaç-Gelendost basin is located in the east of Eğirdir Lake and covers catchments of the Hoyran and Yalvaç rivers which are discharged to the lake. The most important settlements are Gelendost and Yalvaç, and there are many villages and municipalities connected to these districts. Senirkent-Uluborlu basin is located in the northwest of the lake and besides the Senirkent and Uluborlu districts, Büyükkabaca, Küçükabaca, İleydığa, Yassıören, Ortayazı, Garip, Dereköy, Uluğbey, Başköy, and Gençali settlements are located in the basin. According to meteorological data obtained from the State Meteorology Works, the main rainfall varies from 348 to 788 mm in the study area (Şener & Davraz 2013). In recent years, surface and groundwater quality has been under threat in the basin due to many pollution sources, such as open dumps, fertilizer and pesticides practices in agricultural areas, uncontrolled sewerage system, mining activities, etc. (Şener 2010). Agricultural production takes place in approximately 1,147 km² of the Eğirdir Lake basin. In these regions, natural and synthetic fertilizers and agricultural pesticides have been used in large amounts, and this is the main cause of the degradation of groundwater quality in the study area (Şener et al. 2009).

**Geological and hydrogeological settings**

The major factors that control the chemical quality of groundwater are anthropogenic and geologic impacts originating from pollution sources and water–rock interaction in the
basin. Therefore, lithological characteristics of geological units were first identified and a geological map of the basin prepared (Figure 2). The oldest rock unit is the Upper Cambrian aged metamorphic, which consists of low grade meta-sedimentary material including meta-sandstone, meta-siltstone, schist, calc-schist, recrystallized limestone and quartzite as well as limestone, sandstone, and conglomerate affected by very low grade metamorphism in the study area (Demirkol 1982; Özgül et al. 1991). Mesozoic aged flysch overlies the metamorphic units unconformably. The flysch units at the southeast edge of the Eğirdir Lake consist of siliciclastic shales. Autochthonous carbonate rocks are located on a large scale in the east and west of the lake. In general, carbonate rocks consist of limestone and dolomite in the study area. Most of the carbonate rocks are highly karstified. Dolomite consists of white, massive, fine-grained dolomite with frequent supra-tidal textures about 150 m thick. Limestone is mainly in different colors, thin to medium thick bedded micritic limestones (Elitok 2002). Allochthonous ophiolitic complex overlies carbonate units unconformably. The units crop out in the south and northeast of the basin. The complex consists of radiolarian, cherts, diabase, sandstone, mudstone, and limestone. Paleogene aged clastics overlie the ophiolitic units in the west of the study area. Clastics consist of sandstone, claystone, siltstone, and conglomerates, namely flysch. Also, Neogene aged clastic and volcanic units are located in the basin. Clastic units crop out in the east of the basin, namely Neogene deposits. This fluviatile-lacustrine and terrestrial formations unconformably overlie the rock units that had previously existed. Although these units show an upward lithology of conglomerate–sandstone and siltstone–argillaceous limestones–limestone, they also show lateral and
vertical transitions (Demirkol 1982). Volcanic units are located in the west of the lake and overlie carbonates and flysch unconformably. Quaternary deposits are composed of materials such as clay, silt, sand, and gravel, unconformably covering all of the other lithological units (Şener 2010).

Seven hydrogeologic units were determined in the study area based on hydrogeological properties of the lithological units. The Quaternary alluvium and the Mesozoic carbonate rocks are classified as aquifer units in the basin. Pyroclastic units and Neogene deposits were identified as semi-permeable units. In addition, ophiolite complex, flysch, and metamorphic rocks were identified as impermeable units due to having low permeability. Alluvium is the most important aquifer due to the porous structures and groundwater is taken from alluvial aquifer in the basin. According to groundwater level data, the groundwater depth varies from 3 to 36 m in the Senirkent-Uluborlu basin, 1–21.6 m in the Yalvaç basin, and 0.15–51.2 m in the Gelendost basin (Şener & Davraz 2013). The groundwater flow direction of the alluvium aquifer is toward the Eğirdir Lake in the basin. The amount of groundwater discharge to the east side of Eğirdir Lake by means of the karstic aquifer was determined as 114 hm³/year using the MODFLOW model (Soyaslan 2004). Yalvaç and Hoyran streams discharge 26.38 and 3.42 m³/s to Eğirdir Lake, respectively (Şener 2010). In the west of the lake, the amount of groundwater discharge from alluvium aquifer to Eğirdir Lake has been calculated as 7.8 hm³/year (Seyman 2005). Pupa Stream is the most important surface water flowing through the basin and it discharges 12.84 m³/year to Eğirdir Lake (Seyman 2005; Tay 2005). Hydraulic conductivity determined in the west of the study area varies between 8.72 × 10⁻⁶ and 2.24 × 10⁻⁴ m/s in alluvium. Hydraulic conductivity was calculated as 1.18 × 10⁻⁵ to 5.6 × 10⁻⁷ m/s in the east of the basin (Şener & Davraz 2013).

Pollution sources

Groundwater contamination can be classified into two groups: natural (geogenic) and man-made (anthropogenic) sources. Natural groundwater contamination is primarily caused by water–rock interaction, geothermal field effects and/or infiltration from low quality rivers, lakes, or seawater.

Anthropogenic groundwater contamination is generally attributed to the excessive use of: agricultural pesticides and fertilizers; domestic or industrial wastewaters, mining waste products; disposal of industrial waste; waste disposal sites; and poor well construction (Baba & Ayyıldız 2006; Şener et al. 2003). In the study basin, groundwater quality is affected negatively by point and non-point pollution sources such as domestic wastewater, constructed wetlands, uncontrolled landfills, industrial and agricultural activities (Figure 3). There are 81 settlements in the basin, and the total population of these settlements is approximately 175,000 (Günes 2008). The Eğirdir and Yalvaç districts have wastewater treatment plants. However, the wastewater that originates from other settlements comes from scattered areas collecting in septic tanks or discharges to the streams directly. In such case, groundwater contamination is inevitable due to wastewater in the basin. There are nine constructed wetlands (in Senirkent, Uluborlu, Yalvaç, Gelendost, and Eğirdir districts) in the basin. Constructed wetlands are natural wastewater treatment systems which are used for accumulating domestic and industrial wastewaters. However, their capacity is not sufficient for treatment in the present case; on the contrary, they are affected by surface and groundwater quality negatively (Şener et al. 2013). In addition, the solid wastes are scattered in open dump areas because there is no controlled landfill site in the basin and the leachate generating from the open dump areas mix with surface or groundwater. In the basin, the main industrial pollutant is Yalvaç Leather Tanneries. The wastewater of the 62 enterprises belonging to the tanneries is purified in the Yalvaç treatment plant, but sometimes these wastewaters directly flow into the Yalvaç Stream and its drainage canal. The most important non-point pollution source is the agricultural activity observed in the basin, during which, fertilizers (synthetic and natural) and pesticides are used extensively in order to increase product quality and quantity. Farmyard manure is used more intensively than synthetic fertilizer with 95,000 t used annually. Also, annually used amounts of nitrogen (pure N), phosphorus oxide (P₂O₅), and potassium oxide (K₂O) are 2,591, 2,002, and 343 t, respectively (Anonymous 1999; Şener et al. 2013).

Sampling and analytical procedure

Groundwater samples were collected during October 2014 representing the dry season in accordance with United States Environmental Protection Agency methodologies (USEPA
A total of 29 water samples taken from wells/springs and the coordinates of these locations were loaded in the Magellan eXplorist 600 Manual Global Positioning System (GPS). Samples were stored in two polyethylene bottles. One of the bottles was acidified with suprapure HNO₃ for determination of cations and another was kept unacidified for anion analyses. Bottles labeled to avoid misidentification were rinsed in clear spring water several times and then filled to the top to minimize the entrapment of air in water samples (Larsen et al. 2000), then stored at 4 °C in a refrigerator.

The pH, temperature (T, °C), electrical conductivity (EC, μS/cm), total dissolved solids (TDS, mg/L), and dissolved oxygen (DO, mg/L), were measured in situ with YSI Professional Plus handheld multi-parameter instruments that were calibrated with standard solutions. The major chemical constituents were analyzed at the ACME Laboratory (Canada-ISO 9002 Accredited Co.). The major cation (Ca, Mg, K, Na) and trace metal (Al, As, B, Ba, Cr, Cu, Fe, Mn, Ni, Pb, Zn) amounts were determined by inductively coupled plasma mass spectrometry within group 2C-MS in the ACME Laboratory. Chloride, bicarbonate, sulfate, and nitrate analyses were performed in the Eğirdir Fisheries Research Institute Laboratory (Isparta/Turkey). The bicarbonate concentrations were determined by titrimetric method. The argentometric method based on titration of a sample with silver nitrate was used for the determination of chloride (AWWA 1995). Sulfate was determined spectrophotometrically by barium sulfate turbidity method (Clesceri et al. 1998; AOAC 1995). In addition, determination of nitrate was performed by using spectrophotometer reagents and WTW
photoLab Spectral-12 Spektrophotometre. The calculated charge–balance error of the water samples is <5%, and this ratio is within the limits of acceptability.

**Water quality index calculation**

A water quality index (WQI) is a numeric expression used to evaluate the quality of water and to be easily understood by managers (Bordalo et al. 2006). The WQI provides comprehensible information from complex data of water quality for most domestic uses. Also, it can be used for evaluating the influence of natural and anthropogenic activities based on several key parameters of groundwater chemistry (Kumar et al. 2013). WQI is defined as a rating that reflects the composite influence of different water quality parameters (Sahu & Sikdar 2008). First, each of the chemical parameters was assigned a weight \( w_i \) based on their perceived effects on primary health. The highest weight of five was assigned to parameters which have the major effects on water quality. The relative weight \( W_i \) is computed from the following equation:

\[
W_i = \frac{w_i}{\sum_{i=1}^{n} w_i}
\]  

where \( W_i \) is the relative weight, \( w_i \) is the weight of each parameter and \( n \) is the number of parameters. \( \sum w_i \) is the sum of all the parameters. Then, a quality rating scale \( q_i \) for each parameter is assigned by dividing its concentration in each water sample by its respective standard and the result multiplied by 100:

\[
q_i = \frac{C_i}{S_i} \times 100
\]  

where \( q_i \) is the quality rating scale, \( C_i \) is the concentration of each chemical parameter in each water sample in mg/L, and \( S_i \) is the World Health Organization standard for each chemical parameter in milligrams per liter according to the guidelines of the WHO (2008). To calculate WQI, first, \( S_i q_i \) value should be determined with the following equations:

\[
S_i q_i = w_i q_i
\]

\[ WQI = \sum S_i q_i \]  

The computed WQI values are classified into five categories as follows (Sahu & Sikdar 2008; Yidana & Yidana 2010).

- \(<50\): excellent water
- \(50–100\): good water
- \(100–200\): poor water
- \(200–300\): very poor water
- \(>300\): unsuitable for drinking

**Mapping**

The advancement of GIS and spatial analysis helps to integrate the laboratory analysis data with the geographic data and to model the spatial distributions of water quality parameters, most robustly and accurately (Shabbir & Ahmad 2013). In the present study, the distribution maps of the water quality indices were prepared by using GIS techniques. First, the coordinates of the sampling points were determined using a hand-held GPS device and the locations were imported into GIS software through point layer. The database file including sample codes and all analyses results of the chemical parameters was prepared and this geodatabase was used to generate the spatial distribution map of the WQI. For this, ArcGIS software, Spatial Analyst extension, and inverse distance weight (IDW) interpolation methods were applied in the study. IDW is most suitable in the formulation of the interpolation of isodynamic contours. It also produces smooth and continuous surface changes between observations (Mantzafleri 2007).

**RESULTS AND DISCUSSION**

The qualities of a water resource depend on the management of anthropogenic discharges as well as the natural physicochemical characteristics of the catchment areas (Efe et al. 2005). The statistical summary of the physicochemical parameters and limit values for drinking waters is presented in Table 1. The chemical composition of the water samples \((n = 29)\) in the study region shows a wide range. The temperature variation ranges from 10.16 to 20°C with a mean value of
13.7 °C. pH is one of the most important operational water quality parameters, with the optimum pH required often being in the range of 6.5–9.5 (WHO 2008). It is controlled by carbon dioxide, carbonate, and bicarbonate equilibrium (Hem 1985). The minimum and maximum values of pH were measured as 8.2 and 8.68, respectively, with a mean value of 8.4. This shows that the groundwater of the study area is slightly alkaline due to the presence of carbonates and bicarbonates from karstic rocks. EC is a measure of water capacity to convey electric current. The presence of various dissolved salts is responsible for the EC of water. The EC in the study region varied from 175 to 692 μS/cm with an average of 425.90 μS/cm. TDS is a measure of the combined content of all inorganic and organic substances contained in a liquid in molecular, ionized, or microgranular suspended form (Saravanakumar & Ranjithkumar 2011). Concentrations of TDS in water vary considerably in different geological regions owing to differences in the solubilities of minerals (WHO 2004). TDS ranged from 222.2 to 794.9 mg/L with a mean value of 465.1 mg/L. The permissible limit of TDS of drinking water is 500 mg/L (WHO 2008). The observation shows that the TDS exceeded the maximum permissible limit in 12 locations. In addition, Todd (1980) suggested that groundwater be classified using TDS into very fresh (0–250 mg/L), fresh (250–1,000 mg/L), brackish (1,000–10,000 mg/L), and saline (10,000–100,000 mg/L) (Boateng et al. 2016). According to this categorization, all the groundwater samples fell under the fresh water type in the study area. DO is an essential parameter for the survival of all aquatic organisms and oxygen is the most well established

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<td>425.90</td>
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<td>pH</td>
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<td>8.40</td>
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<td>465.1</td>
<td>176.9</td>
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<td>386.25</td>
<td>112.00</td>
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<td>138.67</td>
<td>159.31</td>
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<td>10.76</td>
<td>6.21</td>
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<td>108.31</td>
<td>46.28</td>
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<tr>
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<td>0.0133</td>
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<tr>
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<td>0.0741</td>
<td>0.0137</td>
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</table>
indicator of water quality (Said et al. 2004). The optimum value for good water quality is 4 to 6 mg/L of DO, which ensures healthy aquatic life in a water body (Alam et al. 2007; Avvannavar & Shrihari 2008). The in situ measured DO values of the water samples ranged from 7.1 to 10.7 mg/L with a mean value of 8.9 mg/L.

The major anions abundance in the study area was in the order of $\text{HCO}_3^– > \text{SO}_4^{2–} > \text{Cl}^– > \text{NO}_3^–$. The concentration of carbonates in natural waters is a function of dissolved carbon dioxide, temperature, pH, cations, and other dissolved salts. Bicarbonate is present in considerable amounts according to carbonate ions. Also, bicarbonate concentration of natural waters generally held within a moderate range by the effects of the carbonate equilibrium (Kumar et al. 2015). The bicarbonate concentration varied between 169.58 and 606.34 mg/L with a mean of 386.25 mg/L. The concentration of bicarbonate in the study area was mostly within the WHO (2008) standards except for four samples, Y20, Y21, Y22, and Y24. Sulfate occurs naturally in numerous minerals and is used commercially, principally in the chemical industry (Nas & Berktay 2010). It is one of the least toxic anions, even though dehydrogenation is observed at high concentrations (Varol & Davraz 2015b). According to WHO (2008) and TS-266 (2005) the highest desirable and maximum permissible limit of sulfate is 250 mg/L. The sulfate concentration in the groundwater samples varied between 7.11 and 495.5 mg/L with a mean concentration of 138.67 mg/L. Sulfate exceeded the maximum permissible limit in six locations, Y21, Y23, Y24, Y27, Y28, and Y29. The chloride in groundwater may be from diverse sources such as weathering, leaching of sedimentary rocks and soil, domestic and municipal efﬂuents (SarathPrasanth et al. 2012). No health-based guideline value is proposed for chloride in drinking water. However, high concentrations of chloride give a salty taste to water and beverages (WHO 2004). The concentration of chloride ion in groundwater of the study area varied between 8.87 and 26.95 mg/L with a mean of 13.88 mg/L. All groundwater samples were within the maximum permissible limit of 250 mg/L. Nitrate is the product of nitrogenous material conversion and aerobic stabilization of organic nitrogen (Shabbir & Ahmad 2015). The nitrate concentration in groundwater and surface water is normally low but can reach high levels as a result of leaching or run-off from agricultural and/or contamination from human or animal wastes as a consequence of the oxidation of ammonia and similar sources (WHO 2004). The nitrate concentration ranged from 0.87 to 3.87 mg/L with a mean value of 2.34 mg/L in the study area; the permissible limit of nitrate is given as 50 mg/L by the TS-266 and WHO for drinking water. According to analysis results, all of the groundwater samples were within the maximum permissible limit.

The order of the major cation trend was $\text{Ca}^{2+} > \text{Mg}^{2+} > \text{Na}^+ > \text{K}^+$ and $\text{Ca}^{2+}$ is the dominant ion among the cations in the study area. Calcium and magnesium are directly related to hardness of the water and these ions are the most abundant elements in the surface and groundwater, and exist mainly as bicarbonates and, to a lesser degree, in the form of sulfate and chloride (Kumar et al. 2015). Ca can be derived from dissolution of carbonate minerals (e.g., calcite, dolomite, aragonite) as well as carbonate cement within formations. The primary source of Mg in natural water is ferromagnesian minerals (olivine, diopside, biotite, hornblend) within igneous and metamorphic rocks and magnesium carbonate (dolomite) in sedimentary rock (Singh et al. 2012). The Ca and Mg concentrations of water samples vary from 43.86 to 198.21 mg/L and 5.87 to 46.01 mg/L, respectively. Calcium concentrations of all groundwater samples were within the permitted limit of 300 mg/L. However, magnesium exceeded the maximum permissible limit in ten locations, Y3, Y4, Y8, Y9, Y11, Y12, Y19, Y20, Y21, and Y26. The Na concentration ranged from 1.2 to 25.76 mg/L with a mean value of 10.76 mg/L in the study area; the permissible limit of sodium is given as 200 mg/L by the TSE and WHO for drinking water. According to analysis results, groundwater samples were within the maximum permissible limit. Potassium is a naturally occurring element and the amount of potassium varied between 0.36 and 28.76 mg/L with an average value of 4.13 mg/L. It was found that all the samples with potassium content within the permissible limit, except for sample Y14.

Trace metals are the most persistent and dangerous pollutants in aquatic ecosystems. They disturb the natural balance of ecosystems that have been formed evolutionarily over a long period of time (Arnason & Fletcher 2003; Bai et al. 2011). Trace metal accumulations determined in waters and sediment indicate the presence of natural (geogenic) or anthropogenic sources. Water–rock interaction within drainage basins is the primary source for the lithogenic
contribution of heavy metals into an aquatic system. Also, anthropogenic sources such as urban and industrial waste water, mining and smelting operations, combustion of fossil fuels, processing and manufacturing industries, waste disposal including dumping are primary pollutants for aquatic systems (Pardo et al. 1990; Şener et al. 2014). Al, As, B, Ba, Cr, Cu, Fe, Mn, Ni, Pb, U, and Zn analyses were performed in water samples and it was found that the concentrations of the Ba, Cu, Mn, Ni, Pb and U parameters were within the permissible limit given by the TSE and WHO for drinking water. However, there is no limit value for B and Zn parameters. The minimum, maximum, and mean values of the trace metal parameters can be seen in Table 1. The Al and As contents of water samples were determined as a range 0.001–0.431 mg/L and 0.004–0.0128 mg/L, respectively. According to analysis results, all of the groundwater samples were within the maximum permissible limit in terms of Al except for sample Y8. However, As concentrations were over the permissible limit of the WHO (2008) and TS-266 (2005) in six locations, Y1, Y4, Y5, Y6, Y7, and Y8. The B and Ba contents of water samples were determined as a range 0.009–0.0910 mg/L and 0.005–0.556 mg/L, respectively and all of the groundwater samples were within the maximum permissible limit for Ba. The concentration of Cr ranges from 0.0009 to 0.054 mg/L in groundwater and was mostly within the TSE and WHO standards except for one sample (Y3). The concentration of Cu was within the maximum permissible limits with minimum and maximum of 0.0007 and 0.0341 mg/L, respectively. The Fe concentration in the groundwater samples varied between 0.009 and 0.581 mg/L with an average concentration of 0.0468 mg/L, and the concentration of Fe was within the WHO standards except for only one water sample (Y8). The trace metal results indicated that sample Y8 has high Al, As, and Fe content and sample Y3 has high Cr content. Both of them are located in the Senirkent-Uluborlu basin and the main reason for this excessive dosage is pesticide and fertilizer usage during agricultural activities and also water–rock interaction with volcanic units. The Mn and Ni contents of water samples were determined as a range 0.00004–0.0485 mg/L, and 0.0001–0.0036 mg/L, respectively. In addition, the Pb, U, and Zn contents of water samples were determined as a range 0.0001–0.0035 mg/L, 0.0003–0.0101 mg/L, and 0.0009–0.0741 mg/L, respectively (Table 1).

Hydrochemical types

Water chemistry is mainly influenced by water–rock interaction taking place from the recharge area to sampling location (Purushothaman et al. 2014). In addition, hydrogeochemical types reflect the effects of chemical reactions occurring between the minerals within the lithologic framework and groundwater (Varol & Davraz 2013a). In the present study, the groundwater samples were classified hydrochemically using major cations and anions with conventional Piper trilinear diagram (Piper 1944) to determine the similarities between groundwater in the basin. Also, a Gibbs diagram was used to understand the genesis of groundwater.

Gibbs diagram

Gibbs (1970) suggested that a simple plot of TDS versus the weight ratio of Na⁺/(Na⁺ + Ca²⁺) could provide information on the relative importance of three major natural mechanisms controlling surface water chemistry: (1) atmospheric precipitation; (2) rock weathering; and (3) evaporation and fractional crystallization. Three distinct fields, rainfall dominance, evaporation dominance, and rock–water interaction dominance areas are shown in the Gibbs diagram, also known as Boomerang plot, which is widely employed to assess functional sources of dissolved chemical constituents (Gibbs 1970). The Na⁺, Ca²⁺, and TDS data of the groundwater samples have been plotted in the diagram and Gibbs plot shows that the all sampling points mostly fall in the rock–water interaction dominance zone (Figure 4).

Piper trilinear diagram

The graphical representation of Piper’s diagram (Piper 1944) is extensively used to assess the geochemical evolution in groundwater flow systems. Two triangular fields were plotted separately from the percentage meq/L values of the cations Ca²⁺ and Mg²⁺ (alkaline earths) and Na⁺ (alkali), and the anions HCO₃⁻ (weak acid) and SO₄²⁻ and Cl⁻ (strong acid) and then projected onto the central field for the representation of overall characteristics of water. This plot reveals similarities and differences among groundwater samples and also relationships for large sample groups (Srivastava & Ramanathan 2008). The Piper trilinear diagram was prepared using
Aquachem 3.7 scientific software and results indicate that the Ca-Mg-HCO₃, Ca-HCO₃, Ca-SO₄-HCO₃, and Ca-Mg-HCO₃-SO₄ dominant water types were observed in the study area due to water–rock interaction (Figure 5). The limestone and dolomitic rocks are located on a large scale in the study area and carbonate-rich rocks such as crystalline limestone, dolomitic limestone are the major sources for carbonate weathering. The sulfate might come from the breakdown of organic substances of weathered soils, leachable sulfate from fertilizers and other human influences like sulfuric salts in domestic wastewater (Bahar & Yamamuro 2008; Varol & Davraz 2015). The elevated SO₄ concentrations were related to weathered soils and also pollutants such as fertilizer and pesticide used in agricultural activities according to field observations. Ca can be derived from the dissolution of carbonate minerals (e.g., calcite, dolomite, aragonite) as well as carbonate cement within formations. In addition, the major source of Mg in groundwater is probably Mg-bearing minerals such as dolomite and magnesium sulfate minerals in the study area.

**WQI**

WQI is an important parameter for identifying the water quality and its sustainability for drinking purposes (Magesh et al. 2013; Kumar et al. 2015). In the present study, WQI was computed for 29 groundwater samples. First, important water
quality parameters (pH, TDS, HCO\textsubscript{3}\textsuperscript{-}, Cl\textsuperscript{-}, SO\textsubscript{4}\textsuperscript{2-}, NO\textsubscript{3}\textsuperscript{-}, Ca\textsuperscript{2+}, Mg\textsuperscript{2+}, Na\textsuperscript{+}, As, Cr, Mn, Pb, Ni, and U) were selected and a weight was assigned to each parameter depending upon the effect on human health. In addition, the limit values of the World Health Organization’s guidelines (WHO 2008) were utilized in the calculations (Table 2). The highest weight of 5 was assigned to parameters such as TDS, NO\textsubscript{3}, and As which have the major effects on water quality for especially drinking purposes. HCO\textsubscript{3}\textsuperscript{-}, Ca\textsuperscript{2+}, Mg\textsuperscript{2+}, Pb, Ni, and U have been assigned a weight of 3 taking into consideration their importance in water quality. The relative weights (Wi) are computed for each parameter and results are given in Table 2. The WQI values have been calculated using related equations (Equations (2)–(4)) given in the Methodology section, and WQI results and water types for individual samples are presented in Table 3. WQI interpolation maps of the study area were prepared using GIS techniques and are presented in Figure 6.

The computed WQI for the groundwater samples values ranged from 21.7 to 54.6 in the study and the groundwater quality of the study area is in the ‘excellent’ to ‘good’ range. 89.66% of the groundwater samples represented ‘excellent water’ and 10.34% of the samples fell into ‘good water’ category. According to the WQI distribution map of

### Table 2 | Relative weight of chemical parameters

<table>
<thead>
<tr>
<th>Parameters</th>
<th>WHO (2008) standards</th>
<th>Weight (wi)</th>
<th>Relative weight (Wi)</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>6.5–8.5</td>
<td>4</td>
<td>0.0727</td>
</tr>
<tr>
<td>TDS</td>
<td>500</td>
<td>5</td>
<td>0.0909</td>
</tr>
<tr>
<td>HCO\textsubscript{3}</td>
<td>500</td>
<td>3</td>
<td>0.0545</td>
</tr>
<tr>
<td>Cl</td>
<td>250</td>
<td>4</td>
<td>0.0727</td>
</tr>
<tr>
<td>SO\textsubscript{4}</td>
<td>250</td>
<td>4</td>
<td>0.0727</td>
</tr>
<tr>
<td>NO\textsubscript{3}</td>
<td>50</td>
<td>5</td>
<td>0.0909</td>
</tr>
<tr>
<td>Ca</td>
<td>300</td>
<td>3</td>
<td>0.0545</td>
</tr>
<tr>
<td>Mg</td>
<td>30</td>
<td>3</td>
<td>0.0545</td>
</tr>
<tr>
<td>Na</td>
<td>200</td>
<td>2</td>
<td>0.0364</td>
</tr>
<tr>
<td>As</td>
<td>0.01</td>
<td>5</td>
<td>0.0909</td>
</tr>
<tr>
<td>Cr</td>
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<td>0.0727</td>
</tr>
<tr>
<td>Mn</td>
<td>0.40</td>
<td>4</td>
<td>0.0727</td>
</tr>
<tr>
<td>Pb</td>
<td>0.01</td>
<td>3</td>
<td>0.0545</td>
</tr>
<tr>
<td>Ni</td>
<td>0.07</td>
<td>3</td>
<td>0.0545</td>
</tr>
<tr>
<td>U</td>
<td>0.03</td>
<td>3</td>
<td>0.0545</td>
</tr>
</tbody>
</table>

\[ \sum_{wi} = 55 \quad \sum_{Wi} = 1 \]

The study area, the samples taken from the Yalvaç-Gelendost basin have lower WQI values compared with samples taken from the west of the lake. Domestic, agricultural, and/or industrial pollutants are the main reason for the poor water quality in the Yalvaç-Gelendost basin.

### Risk assessment on human health

A human health risk assessment is the process to estimate the nature and probability of adverse health effects in humans who may be exposed to chemicals in contaminated
environmental media, now or in the future. On a global scale, pathogenic contamination of drinking water poses the most significant health risk to humans. However, significant risks to human health may also result from exposure to non-pathogenic, toxic contaminants that are often ubiquitous in water. In this section, the hazard index (HI) method was used to assess the overall potential for non-carcinogenic effects of metals and nitrate contaminants in groundwater in the Egirdir Lake basin. To assess the overall potential for non-carcinogenic effects posed by more than one chemical, an HI approach is developed based on USEPA (1986a) Guidelines for Health Risk Assessment of Chemical Mixtures.

Hazard identification is to estimate the hazardous effect of the contaminant. In the first step, the hazard level is examined by physical and chemical properties of contaminants such as mobility and contaminant levels at the point of exposure where the contaminants are exposed to the environment. The second step is exposure assessment, estimated by average daily dose (ADD) using the identification of intensity, frequency, exposure period, and pathway of contaminants. In the third step, dose–response assessment examines the relationship between adverse effects and exposure levels of carcinogenic and non-carcinogenic chemicals. The two principal toxicity indices are known as $SF$ (cancer slope factor) and reference dose ($RfD$). The $SF$ and $RfD$ values can be obtained from the EPA Integrated Risk Information System (IRIS) on-line database and EPA Health Effects Assessment Summary Tables (USEPA 1994). Risk characterization is the final step that predicts the level of risk. The results of exposure assessment and dose–response assessment are integrated to derive quantitative estimates of cancer risk and HI (USEPA 1986a, 1986b, 1989; Lee et al. 2006).

Exposure of human beings to metals could occur via three main pathways, direct ingestion, inhalation through

Figure 6 | Spatial distribution map of the WQI.
mouth and nose, and dermal absorption through exposed skin, while ingestion and dermal absorption are common for drinking water (USEPA 2004; De Miguel et al. 2007; Wu et al. 2009; Li & Zhang 2010). The dose received through the individual pathway considered was determined using Equations (5) and (6) modified from the US Environmental Protection Agency (USEPA 2004).

\[
\text{ADD}_{\text{oral ingestion}} = \frac{C_i \times L \times EF \times ED}{BW \times AT} \tag{5}
\]

\[
\text{ADD}_{\text{dermal}} = \frac{C_i \times SA \times Kp \times ET \times EF \times ED \times 10^{-3}}{BW \times AT} \tag{6}
\]

where \(C_i\) is the concentration of pollutant \(i\) in drinking water (mg/L); \(L\) is the daily water ingestion rate (L/day); \(EF\) is the exposure frequency (days/year); \(ED\) is the exposure duration (year), taken as 30 years for non-carcinogens and 70 years for carcinogens; \(BW\) is the bodyweight (kg); \(AT\) is the average exposure time (in day), 30 years \(\times\) 365 days/year for non-carcinogens and 70 years \(\times\) 365 days/year for carcinogens; \(SA\) exposed skin area, unit in cm\(^2\); \(Kp\) dermal permeability coefficient in water, unit in cm/h; \(ET\) exposure time, unit in h/day.

The default values used to estimate potential exposure from drinking contaminated water and the default values (USEPA 2001) that are used to estimate dermal ADD for adults and children are given in Table 4. \(Kp\) is dermal permeability coefficient in water, unit in cm/h. The default permeability constants for all other inorganic compounds are provided in USEPA (2004) and \(Kp\) values for several inorganic compounds are given in Table 5.

Risk characterization was quantified by carcinogenic risk and non-carcinogenic risk. Potential non-carcinogenic risks, reflected by the hazard quotient \((HQ)\), were estimated by comparing exposure or average intake of contaminants from each exposure route (ingestion, dermal) with the corresponding \(RfD\) using Equations (7) and (8). If the \(HQ\) exceeds 1 \((HQ > 1)\), there might be concern about non-carcinogenic effects. The higher the value the greater likelihood of adverse non-carcinogenic health effect (USEPA 1989; Khan et al. 2008; Muhammad et al. 2011; Qaiyum et al. 2011; Jamaludin et al. 2013; Equation (3)). To evaluate the total potential non-carcinogenic risks posed by more than one pathway, the \(HI\) was introduced, which was the sum of the \(HQs\) from all applicable pathways. \(HI > 1\) indicated the potential for an adverse effect on human health or the necessity for further study (USEPA 2001, 2004):

\[
\text{Hazard Quotient} (HQ) = \frac{\text{ADD}}{\text{RfD}} \tag{7}
\]

where \(RfD\) is the reference dose (mg/kg/d), and given in Table 5. \(RfD\) values employed in this study were obtained from USEPA (IRIS 2005; Kavcar et al. 2009; Li & Zhang 2010).

Carcinogenic risk is the probability of an individual developing any type of cancer from lifetime exposure to carcinogenic hazards. The acceptable or tolerable risk for regulatory purposes is in the range of \(10^{-6}\) to \(10^{-4}\) (Rodriguez-Proteau & Grant 2005; Lim et al. 2008; Li & Zhang 2010). The following is the basic equation used for lifetime cancer risk assessment of pollutant in drinking water:

\[
\text{cancer risk} = \text{ADD} \times SF \tag{8}
\]

where cancer risk is the carcinogenic risk of pollutant in drinking water (unitless); \(SF\) is the slope factor for pollutant (kg d/mg); \(ADD\) is the chronic daily intake (mg/kg d) for pollutant (Li et al. 2007; Liu et al. 2009). A \(SF\) is most commonly used to evaluate potential human carcinogenic risks (USEPA 1989). \(SF\) values (Table 5) are provided in the IRIS on the website of USEPA (2013).

Table 5 presents the \(HQ\), \(HI\), and \(cancer risk\) values for the oral and dermal pathways relating to adults and children, respectively. \(HQ_{\text{ingestion}}\) (hazard index by oral ingestion) of all
<table>
<thead>
<tr>
<th>Element</th>
<th>Kp (cm/h)</th>
<th>RFD-oral (mg/kg/d)</th>
<th>RFD-dermal (mg/kg/d)</th>
<th>SF (kg d/ mg)</th>
<th>HQ\textsubscript{Ingestion}</th>
<th>HQ\textsubscript{Dermal}</th>
<th>HI – \Sigma HQs</th>
<th>Cancer risk\textsubscript{Ingestion}</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Adult</td>
<td>Child</td>
<td>Adult</td>
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</tr>
<tr>
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<td>$1.00 \times 10^{-2}$</td>
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<td>$1.22 \times 10^{0}$</td>
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<td>$2.08 \times 10^{-2}$</td>
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</tr>
<tr>
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<td>1.4</td>
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<td>$1.00 \times 10^{-2}$</td>
<td>$1.01 \times 10^{-3}$</td>
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<tr>
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<td>$9.14 \times 10^{-3}$</td>
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<td>$1.07 \times 10^{-3}$</td>
<td>$1.09 \times 10^{-3}$</td>
<td>$1.63 \times 10^{-3}$</td>
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<td>0.8</td>
<td>$6.52 \times 10^{-2}$</td>
<td>$1.01 \times 10^{-2}$</td>
<td>$1.01 \times 10^{-2}$</td>
<td>$1.10 \times 10^{-2}$</td>
<td>$1.62 \times 10^{-2}$</td>
<td>$1.02 \times 10^{-2}$</td>
</tr>
</tbody>
</table>
trace elements and nitrate except As for adults was less than 1 in all sample locations, suggesting that these elements posed little hazard. $HQ_{\text{ingestion}}$ of As was more than 1 implying that As may cause adverse health effects and potential non-carcinogenic concerns. $HQ_{\text{ingestion}}$ of As for adults was more than 1 in three locations and was near unity in other locations (Figure 7). However, $HQ_{\text{ingestion}}$ of As for child was more than 1 in all of the locations except for two samples in the Eğirdir Lake basin, indicating serious health concerns (Figure 8). In addition, $HQ_{\text{ingestion}}$ of Cr for child was more than 1 at only one location. The $HQ_{\text{dermal}}$ (hazard index by dermal absorption) of all trace elements and nitrate for adults and children was below unity, indicating that these metals posed little hazard via dermal absorption. The largest value of $HQ_{\text{dermal}}$ was 0.15, which was for Cr for child. Overall HI of As for child exceeded 1, and HI of As for adults also exceeded 1 in seven locations (Y1, Y4, Y5, Y6, Y7, Y8, Y14). HI of Cr for child was more than 1 in Y3 location and HI of Cr for adults was near 1 in the same location. It can be concluded that the highest contributors to chronic risks were As and Cr for both adults and children. This indicated that As posed serious health concerns to the local residents via oral intake, while other metals via oral intake and all the elements via dermal absorption had no or little health threat.

Carcinogenic risk of As through oral intake for child exceeded the target risk of $1 \times 10^{-4}$ (Table 5) and indicated that the ingestion of water over a long lifetime could increase the probability of cancer. The risk assessment indicated that As was the most important pollutant in the Eğirdir Lake basin. Previous studies reported adverse health effects including hypertension, neuropathy, diabetes, skin lesions, and cardiovascular diseases through high arsenic intake (Avani & Rao 2007; Bhattacharya et al. 2007; Wu et al. 2009). Therefore, special attention should be paid to arsenic for local residents, particularly for sensitive children, and measures need to be taken to sustain a healthy aquatic ecosystem.

Figure 7 | Spatial distribution map of $HQ_{\text{ingestion}}$ for As (adults).
CONCLUSIONS

The hydrochemical characteristics, groundwater quality, and human health risk in Eğirdir Lake basin was evaluated in the present study. Eğirdir Lake is an indispensable water source for the region due to usage aims such as drinking/irrigation water, tourism, and fishing. The groundwater is used as drinking and irrigation water in the study area. A total of 29 samples were taken from wells within the study area and analyzed for hydrochemical and quality evaluation. The order of anion and cations are \( \text{HCO}_3^- > \text{SO}_4^{2-} > \text{Cl}^- > \text{NO}_3^- \) and \( \text{Ca}^{2+} > \text{Mg}^{2+} > \text{Na}^+ > \text{K}^+ \) in groundwater samples and \( \text{HCO}_3^- \) and \( \text{Ca}^{2+} \) are the dominant ions among the anions and cations in the study area. According to Piper trilinear diagram, \( \text{Ca-Mg-HCO}_3^-, \text{Ca-HCO}_3^-, \text{Ca-SO}_4^2-, \text{HCO}_3^-, \text{Ca-Mg-HCO}_3^2-\text{SO}_4^- \) are the dominant water types related to water–rock interaction. Carbonate weathering plays an active role in development of the water type. Also, Gibbs plot indicates that all the samples fall in the rock–water interaction dominance zone. The results of the analyses were compared with drinking water limit values determined by WHO (2008) and TS-266 (2005) to assess the potability of groundwater. The results show that \( \text{HCO}_3^-, \text{SO}_4^{2-}, \text{Mg}^{2+}, \text{Al}, \text{As}, \text{Cr}, \text{and Fe} \) are a little over the WHO (2008) and TS-266 (2005) limit values. All the other parameters are within the permissible limit for drinking water. In the study area, groundwater quality is slowly reaching an unsuitable stage for drinking water due to industrial and agricultural activities. Moreover, water–rock interaction affects the water quality adversely. According to the WQI classification, the water samples fall into the excellent to good water category. In general, groundwater quality in Yalvaç-Gelendost basin is lower than Senirkent-Uluborlu basin. However, high Al, As, and Fe content was determined in Senirkent-Uluborlu basin related to water–rock interaction and agricultural activities.

Figure 8 | Spatial distribution map of HQ ingestion for As (children).
Risk assessment is an attempt to identify and quantify potential risks to human health resulting from exposure to various contaminants. In this study, oral ingestion and dermal route were taken into consideration for adults and children. HQ\textsubscript{ingestion} of As for adults was more than 1 in three locations. However, HQ\textsubscript{ingestion} of As for children was more than 1 in all of the locations except for two samples in the Eğirdir Lake basin, indicating serious health concerns. In addition, HQ\textsubscript{ingestion} of Cr for child was more than 1 at only one location. It can be concluded that the highest contributors to chronic risks were As and Cr for both adults and children. This indicated that As posed serious health concerns for local residents via oral intake, while other metals via oral intake and all the elements via dermal absorption posed no or little health threat.

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