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ORIGINAL PAPER

Sobol sensitivity analysis for risk assessment of uranium in groundwater

Deepak Kumar · **Anshuman Singh** · **Pappu Kumar** · **Rishi Kumar Jha** · **Sunil Kumar Sahoo** · **Vivekanand Jha**

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Abstract The exposure to uranium (U) in the natural environment is primarily through ingestion (eating contaminated food and drinking water) and dermal (skin contact with U powders/wastes) pathways. This study focuses on the dose assessment for different age-groups using the USEPA model. A total of 156 drinking water samples were tested to know U level in the groundwater of the study region. Different age-groups were selected to determine the human health impact due to uranium exposure in the residing populations. To determine the relative importance of each input, a variance decomposition technique, i.e., Sobol sensitivity analysis, was used. Furthermore, different sample sizes were tested to obtain the optimal Sobol sensitivity indices. Three types of effects were evaluated: first-order effect (FOE), second-order effect (SOE) and total effect. The result of analysis revealed that 17% of the samples had U concentration above 30 μg l^{-1} of U, which is the recommended level by World Health Organization. The mean hazard index (HI) value for younger age-group was found to be less than 1, whereas the 95th percentile value of HI value exceeded for both age-groups. The mean annual effective dose of U for adults was found to be slightly higher than the recommended level of 0.1 m Sv $year^{-1}$. This result signified that adults experienced relatively higher exposure dose than the children in this region. Sobol sensitivity analysis of FOE showed that the concentration of uranium (C_w) is the most sensitive input followed by intake rate (IR) and exposure frequency. Moreover, the value of SOE revealed that interaction effect of $C_w - \mathrm{IR}$ is the most sensitive input parameter for the assessment of oral health risk. On the other hand, dermal model showed $C_w - F$ as the most sensitive interaction input. The larger value of SOE was also recorded for older age-group than for the younger group.

Keywords Groundwater · Sobol sensitivity analysis · Uranium · Dose assessment · Global sensitivity analysis

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D. Kumar (✉) · A. Singh · P. Kumar
Department of Civil Engineering, National Institute of Technology Patna, Bihar 800005, India
e-mail: decage007@gmail.com

R. K. Jha
Department of Mathematics, National Institute of Technology Patna, Bihar 800005, India

S. K. Sahoo · V. Jha
Environmental Assessment Division, Bhabha Atomic Research Centre, Trombay, Mumbai 400085, India

Introduction

Low concentration of uranium (U) is ubiquitous and exists within different rocks, soils and water. In natural environment, three isotope mixtures of U are commonly found: ^{238}U (99.27%), ^{235}U (0.72%) and ^{234}U (0.0054%) (Smedley et al. 2006). Other isotopes of U, ^{236}U and ^{233}U also exist as a result of various anthropogenic perturbations. U is a well-known pollutant with its dual characteristics (metallic and radioactive), and its chronic exposure results in potential risks to the human health (Kumar et al. 2018a; Hakonson-Hayes et al. 2002; Dewar 2019; Thorne 2020). High levels of U in soil and groundwater can be considered hazardous to the public health as they get transferred to plants along with the minerals and other nutrients through the root system during plant metabolic activities, growth and development (Asaduzzaman et al. 2015). Therefore, apart from inhalation, injection and skin adsorption, soil plant-food routes are the common channels of radionuclides' entry into human body (Arogunjo et al. 2009). Some epidemiological and toxicological studies showed interlink of U exposure via drinking water and its nephrotoxic effects (Zamora et al. 1998; Kurttio et al. 2002; Pinney et al. 2003), renal effects (tubular cell of kidney) (Arzuaga et al. 2010; Kurttio et al. 2002, 2006a), bone effects (Kurttio et al. 2005) and even cancer in urinary organs (Kurttio et al. 2006b). High concentrations of U in groundwater may be from geogenic sources (i.e., mobilization of U from rock and minerals) and caused by anthropogenic activities (e.g., U mining, fuel fabrication nuclear, and phosphate fertilizer application) (Abdelouas 2006; Guo et al. 2016; Alam and Cheng 2014; Wetterlind et al. 2012). In groundwater system, U mobility depends upon the geochemical process, aquifer materials, dissolution of minerals, sorption, leaching of adsorbed U from the mineral surfaces and groundwater residence time. Furthermore, these geochemical processes are controlled by thermodynamic parameters (i.e., pH, redox, ligand concentrations, temperature, PCO_2, and ionic strength) in the aquifers (Chabaux et al. 2003; Fox et al. 2006; Liu et al. 2009; Liesch et al. 2015). U consists of two predominant oxidation states in that U(VI) is mobile relative to U(IV) in the groundwater systems. U(VI) in aquifers, as U(VI) species may be in groundwater and sediments both. Besides adsorption and desorption, U(VI) reduction is also responsible for its transport. In addition, concentrations and molar ratios of Ca^{2+} and HCO_3^- may affect U(VI) speciation and thus adsorption and desorption as well (Davis et al. 2004).

U exists in environment in various physicochemical as complexes (e.g., uranyl carbonate, uranyl phosphate, uranyl fulvate, and uranyl humate), because carbonate and phosphate are not inorganic ligands (Choppin et al. 2002). The understanding of U geochemistry is very complex issue as it depends upon the reactive transport various multispecies with effect of precipitation, dilution, sorption influenced by various co-solute compositions (e.g., Na^+/Ca^{2+}) and complexation of the groundwater (Moon et al. 2007; Senko et al. 2002, 2005; Mehta et al. 2014).

Experimental and epidemiological research findings have significantly contributed toward the understanding of biokinetics of U in mammalian body, and several models have been proposed and applied successfully (Lipsztein 1982; Durbin 1984; Wrenn et al. 1983; Fisher et al. 1991; Leggett and Harrison 1995; Leggett and Pellmar 2003) to understand health effect due to U exposure. In recent times, risk assessors and regulators have gained focus on recognizing influential parameters through uncertainty analysis, and researchers have started avoiding single/point value of input parameters to estimate the risk assessment due to pollutants (Burmaster and Lehr 1991). Such an approach is conservative and results in incremental lifetime risk at the extreme tail end of the risk distribution (Burmaster 1991). Generally, most of the risk assessment models rely on the simulation, i.e., Latin hypercube method and Monte Carlo simulations (Burmaster 1991; Smith 1994), to estimate the most influential parameters and to see their effect. Sensitivity analysis (SA) methods are categorized into two groups: global sensitivity analysis (GSA) and local sensitivity analysis (LSA) methods. In LSA, the partial derivative of the model response is evaluated at local point within the input space at point where derivatives are calculated. It is only suitable for linear model. Interested readers can find a complete review of SA techniques in Saltelli et al. (2008). On the other hand, GSA considers complete input space that gives more generality of the model, and resulting information is independent of the nature of the model. The sensitivity of any inputs in the model is strongly affected by the quality and quantity of the data. In the case of human health risk assessment, it becomes more important as error and uncertainty occur during the whole process (from sampling to end of analysis) of estimation. The occurrence of inherent error can be minimized but cannot be removed completely. To overcome such effects and minimize the inherent uncertainty of input parameters, a variance decomposition-based approach is adopted in this study.

This research adopts Sobol sensitivity analysis to identify the most influential parameters for human risk assessment for U in drinking water using USEPA model. Furthermore, the effect of different sample sizes is also tested to observe behavior-influencing parameters.

Study area and data used

Bihar is situated in the middle Gangetic Plain of India. Figure 1 shows the study area under present investigation. Gaya District falls under Punpun sub-river basin that lies between North latitudes 24°30′: 25° 06′ and East longitudes 84° 24′: 85° 30′, which covers an area of 4976 sq km.

Gaya District was divided into grid pattern of size 6×5 km^2 to cover the entire area for systematic sampling (Kumar et al. 2018b). Considering one representative sample from each grid based on inhabitant populations, a total of 156 drinking water samples were collected. The latitude and longitude of each sample site were identified using Global Positioning System (GPS) (model: Garmin GPS 72H). Prior to sample collection, each source was kept running for 2–5 min as flushing time. Water samples were collected in one-liter low-density polyethylene bottles which were cleaned with 5 M (molarity) of nitric acid after soaking and rinsed with tap water followed by ultrapure distilled water (Milli-Q Ultrapure Water Purification System, Model Z00QSVC01). Samples were collected during the pre-monsoon season in 2017. In order to minimize the wall deposition, all the collected samples were acidified with 65% nitric acid (Suprapur Merck) after filtration (0.45 syringe filters). To determine the U level in water

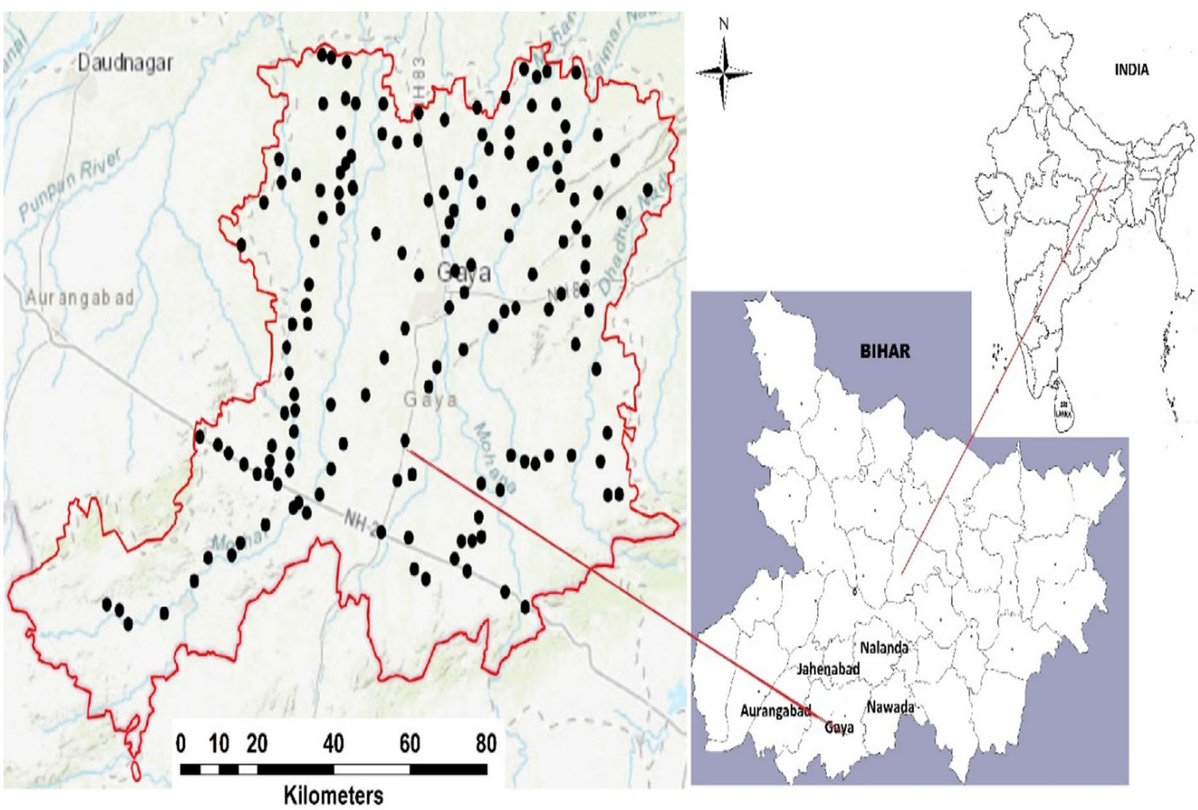

Fig. 1 Study area showing sampling location

samples, laser fluorimetry was used. The details of laser fluorimeter, procedure for analysis and quality control can be found in Kumar et al. (2018a, b).

Methodology

Sobol sensitivity analysis (SSA)

SSA method belongs to the group of GSA, in which all the model inputs are varied simultaneously over the complete input range (Saltelli et al. 1999; Sobol 1993). The variance decomposition nature of SSA estimates the relative contribution of each influential input and its interaction to the model output in terms of their variance. The contribution of inputs to the model output and their interaction can be estimated using Eq. 1:

$$D(f) = \sum_{i} D_i + \sum_{i<j} D_{ij} + \sum_{i<j<k} D_{ijk} + D_{12\ldots p}, \tag{1}$$

where $D(f)$ denotes the total variance or total effect to the model output, D_i is the first-order variance contribution and D_{ij} is the second order, which is also called interaction between the input parameters. Moreover, the other terms present in Eq. 1 denote the higher order of interactions up to P (total input parameters of the model). First-order and total sensitivity indices are defined as follows:

$$\begin{aligned} &\text{First} - \text{order index (FOI)}: S_i = \frac{D_i}{D} \\ &\text{Total} - \text{order index (TOI)}: S_{\mathrm{Ti}} = 1 - \frac{D_i}{D} \end{aligned} \tag{2}$$

The benefit of adopting SSA method is that it does contain underlying assumption between inputs and model output within the full range input parameters and for their interaction. A systematic procedure to perform this SSA is shown in Fig. 2.

Let us consider the following model shown in Eq. 3:

$$Y = f(X_1, \ldots, Xp) \tag{3}$$

The input variables $X_1, \ldots, Xp$ represent the input variables having known probability distribution (PD), and the model output is represented by Y which is a scalar. The decomposition nature of Sobol method (SM) converts output variance into contributed input factor. To estimate the relative influence of an input factor on the model output, the X_i is fixed as x_i*, considering conditional variance as shown in Eq. 4:

$$V_{X-i}(Y|X_i = x*_i) \tag{4}$$

The variance is calculated over input parameter space P−1, where X_{-i}, denotes all the inputs except X_i. The law of total variance is the basic framework of Sobol method, which can be clearly seen in Eq. (5):

$$V(Y) = V_{X_i}(E_{X_{-i}}(Y|X_i)) + E_{X_i}(V_{X_{-i}}(Y|X_i)) \tag{5}$$

After normalization,

$$1 = \frac{V_{X_i}(E_{X_{-i}}(Y|X_i))}{V(Y)} + \frac{E_{X_i}(V_{X_{-i}}(Y|X_i))}{V(Y)} \tag{6}$$

The first term of Eq. 6 represents the FOE for the parameter X_i., i.e.,

$$S_i = \frac{V_{X_i}(E_{X_{-i}}(Y|X_i))}{V(Y)} \tag{7}$$

The second term of Eq. 7 denotes second order, i.e., SOE. The SOE represents the interaction of the factors (X_i and X_j) contributed to model output Y (i.e., sensitivity to X_i and X_j not expressed in V_i nor V_j) as in Eq. 8. The higher order of interaction (third, fourth and so on) can be estimated through further decomposition.

$$S_{ij} = \frac{V_{ij}}{V(Y)} \tag{8}$$

The total effect (TE) in terms of total-order sensitivity (ST_i) is defined as total contribution of input variance, i.e., summation of first-order factor to $P-1$ factor variance (Homma and Saltelli (1996) that can be estimated using Eqs. 9. Details on this methodology can be found in (Sobol 1993; Saltelli et al. 1999; Zhang et al. 2015; Kumar et al. 2019)

$$\begin{aligned} \mathrm{ST}_i &= \sum_{k\#i} S_k \\ \mathrm{ST}_i &= 1 - \frac{V_{X_{i-1}}(E_{X_i}(Y|X_{-i}))}{V(Y)} \\ \mathrm{ST}_i &= 1 - \frac{E_{X_{-i}}(V_{X_i}(Y|X_{-i}))}{V(Y)} \end{aligned} \tag{9}$$

The following property can be deduced to $0 \le S_i \le \mathrm{ST}_i \le 1$.

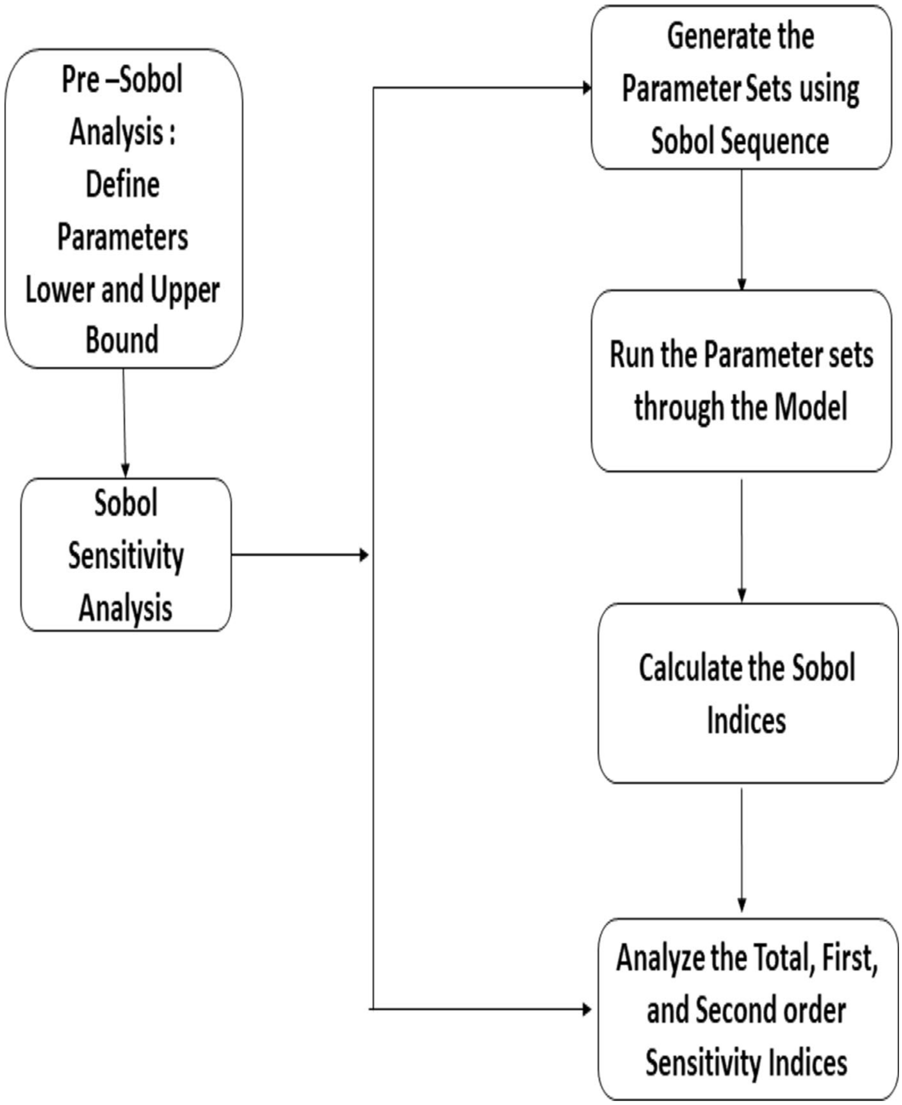

Fig. 2 A systematic flow chart of the SSA (Zhang et al. 2015)

Human health risk model

In this study, two age-groups (children and adults) are selected to evaluate the non-carcinogenic risk (NCR). Daily exposure to U via ingestion and dermal pathways is calculated using Eqs. (10) and (11) introduced by USEPA (USEPA 1992).

$$\mathrm{LADD_{oral}} = \frac{C_w \times \mathrm{IR}_w \times \mathrm{EF} \times \mathrm{ED}}{\mathrm{BW} \times \mathrm{AT}} \quad (10)$$

$$\mathrm{LADD_{derm}} = \frac{C_w \times \mathrm{SA} \times K_p \times F \times \mathrm{ET_S} \times \mathrm{EF} \times \mathrm{ED} \times \mathrm{CF}}{\mathrm{BW} \times \mathrm{AT}} \quad (11)$$

where C_w is concentration (mg l^{-1}) of pollutant in drinking water, LADD lifetime average daily dose intake through drinking water, IR intake rate of water (l day^{-1}), BW body weight (kg), $\mathrm{LADD_{derm}}$ LADD received through skin absorption (mg kg^{-1} day^{-1}), ED exposure duration (years), AT averaging time (Days), SA surface area of skin to get exposure (Cm^2),F fraction of contact surface within skin and water, $\mathrm{ET_s}$ exposure duration during shower (h day^{-1}), EF exposure frequency (day year^{-1}) and *Kp* coefficient of permeation (cm h^{-1}).

The NCR is estimated due to intake of groundwater in terms of hazard quotient (HQ) for both age-groups using Eq. (12):

$$HQ = \frac{\text{LADD or ADD}_{\text{derm}}}{\text{R}f\text{D}} \quad (12)$$

where R*f*D = reference dose (RD) (µg kg^{-1} day^{-1}). Considering the USEPA's Integrated Risk Information System (IRIS), R*f*D for oral ingestion via drinking = 4.48 µg kg^{-1} day^{-1} for U. The value of reference dose via skin exposure is estimated using Eq. (13) recommended by USEPA (Staff 2001).

$$\text{R}f\text{D}_{\text{derm}} = \text{R}f\text{D}_{\text{w}} \times \text{ABS}_{\text{gi}} \quad (13)$$

where $\text{R}f\text{D}_{\text{derm}}$ and $\text{R}f\text{D}_{\text{w}}$ denote the dermal and water RD, respectively, and ABS_{gi} a constant (digestive absorption factor).

The overall NCR is estimated in terms of hazard index (HI), which is calculated using (Eq. 14):

$$\text{HI} = \text{HQ}_{\text{overall}} = \text{HQ}_{\text{w}} + \text{HQ}_{\text{derm}} \quad (14)$$

Results and discussion

Ingestion dose evaluation of uranium

Descriptive statistics of estimated uranium in groundwater samples are shown in Table 1. From the analysis of results, it is clear that mean and median of the U concentration of the samples were well below the recommended levels of the Atomic Energy Regulatory Board (AERB) and the World Health Organization (AERB 2004; WHO 2004) in drinking water.

The NCR value for U via drinking water is calculated for the populations of the studied region. Two age-groups were selected to calculate exposure dose in terms of annual effective dose (AED). The value of AED for adults ranged between 0.3 and 1824 µSv year^{-1}, whereas for children it ranged between 1.18 and 1170 µ Sv year^{-1}. The average value of AED was observed relatively higher for the older age-groups (144.9 ± 237.2) than for the younger groups (92.9 ± 152.0). The mean value and 95th percentile of exposure dose were also found to be greater for adults than for the children. The hazard index (HI) values were also estimated for both oral and dermal exposures considering two age-groups (children and adults). The average of HI was found to be less than 1 for both the age-groups, children (HI = 0.54 ± 0.88) and adults (0.30 ± 0.49), whereas the 95th percentile exceeded the value of 1 in both the cases. This result indicates that children of the study region experienced slightly higher exposure of U than adults. Based on the above results, it was found that study region is slightly prone to the NCR especially to the younger age-groups. Therefore, it is recommended that further water monitoring and dose assessment should be carried out for this region to prevent NCR due to U in drinking water.

The first order and total sensitivity index

The SSA using USEPA model was performed to assess the most influential input associated while calculating the NCR to the population via exposure of U present in drinking water. In order to see the effects of each input of model, different sample sizes ranging from 10 to 14,000 were tested to assure convergence of all the Sobol scores. A Saltelli's sampling scheme was used with 10,000 sample size (N) which resulted in 100,000 (N * (2D + 2)) rows for ingestion evaluation, whereas 12,000 sample size amounted to 168,000 (N * (2D + 2)) rows for dermal model, where D represents the number of input variables (D = 4 for ingestion (oral) and D = 6 for dermal). The final convergence of model inputs was selected based on the highest total

Table 1 Descriptive statistics of uranium (µg l^{-1}) in drinking water of Gaya District of Bihar

District (no. of samples)	Mean	Median	P_5	P_{25}	P_{75}	P_{95}	Min	Max	Skewness	Kurtosis	GM
Gaya (156)	18.9	9.6	0.5	5.1	19.9	64.1	0.1	238.2	4.7	26.9	8.7

Sobol score. Three different types of effect were estimated: FOE, SOE and TE for ingestion and dermal model based on Eq. (12). The model outputs were significantly influenced by the range of input values. Therefore, this study used range values as in Table 2 for sensitivity analysis. The ingestion model had four inputs (concentration of uranium in groundwater (C_w), intake rate (IR), exposure frequency (EF), body weight (BW)), whereas dermal model consisted of six inputs (C_w, F (fraction of contact with skin and pollutant in water), ETS (exposure time during shower), EF, BW, SA (skin surface area)) to evaluate the HI value with three scores of TE, FOE and SOE. The interactive indices value denotes the difference between total order (TO) and first order (FO). The input parameters with value greater than 0.1 are assigned as highly sensitive, whereas sensitivity indices between 0.01 and 0.1 are considered as sensitive (Tang et al. 2006; Wan et al. 2015; Zhan et al. 2013). Moreover, sensitivity indices less than 0.01 are taken as insensitive (Zhan et al. 2013).

Figure 3 represents the Sobol scores of all input variables for two age-groups at different sample sizes for ingestion model. Results of first-order (S1) and total-order (ST) sensitivity indices showed that C_w is the highly sensitive (> 0.1) input parameter. The value of C_w for ST and S1 was found to be higher in the case of adults than in the children over different sample sizes. On the other hand, the values IR for ST and S1 were observed higher for children than for adults. This indicated that children have a much higher effect in terms IR of drinking water. The value of ST and S1 shows that EF is relatively less sensitive than C_w and IR. Moreover, ST value was also observed higher in the case of adults than in children. The Sobol score value of less than 0.01 for BW shows that it is an insensitive parameter for ingestion model. The convergence of the first-order sensitivity indices started at 1000 sample size, and the optimal value was found at 10,000 sample size. Considering C_w as a benchmark parameter (as it was found to be highly sensitive) at 10,000, sample size (N), other indices were also estimated (Figs. 3, 4). The values of FOE and TOE were estimated for both the age-groups. For the children, FOE was recorded highest for $C_w = 69.94$, followed by IR = 14.62, EF = 6.93 and BW = 0.57. On the other hand, for adults, the values were found as follows: $C_w = 78.12$, IR = 8.47 and EF = 7.78. From the result, it was concluded that the older age-groups have higher impact of these input variables while evaluating the ingestion dose (Fig. 5). Moreover, it also showed that BW had no relative importance in

Table 2 Parameter values used for sensitivity for risk assessment HQ model (EPA 2011; WHO 2004; USEPA 1992; Huang et al. 2017)

Parameters	Unit	Population Value ± SD	
		Children	Adults
Water intake rate (IR_w)	$L.day^{-1}$	1.25 ± 0.57	1.95 ± 0.64
Average time (AT)	Days	2190	9125
Exposure frequency (EF)	Day $year^{-1}$	Min:180, max:345, mode:365	Min: 180, max: 345, mode: 365
Exposure duration (ED)	Year	6	70
Body weight (BW)	kg	16.68 ± 1.48	57.03 ± 1.10
Skin surface area (SA)	cm^2	7422 ± 1.25	18,182 ± 1.10
Dermal permeability (Kp)	Cm h^{-1}	1×10^{-3}	1×10^{-3}
Exposure time during shower (ETs)	h day^{-1}	0.13 ± 0.0085	0.13 ± 0.0085
Fraction of skin in contact with water (F)	Unit less	Min:0.4 Max:0.9	Min:0.4 Max:0.9
Fraction of uranium absorbed in gastrointestinal tract (ABS_{gi})	Unit less	1	1
Oral reference dose (RfD_0)	μg kg^{-1} day^{-1}	4.48	4.48

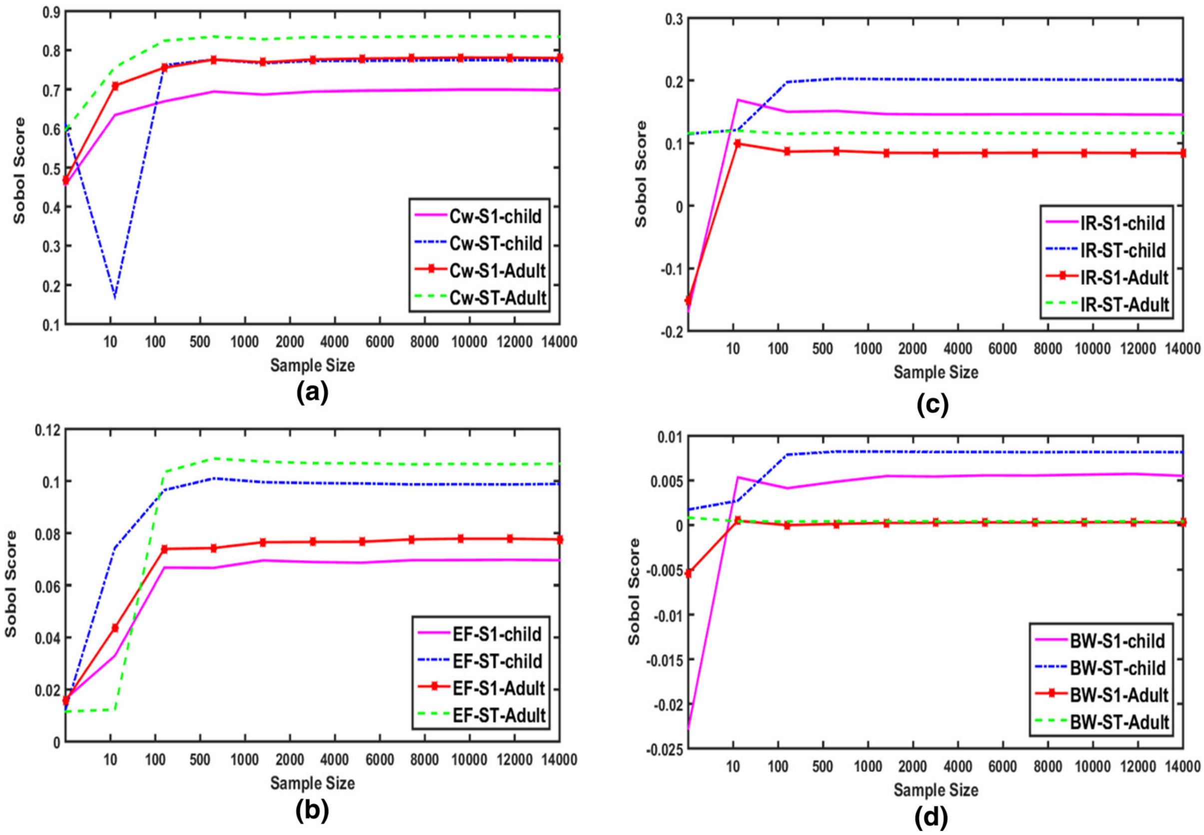

Fig. 3 Evolution of the first-order (S1) and total-order (ST) sensitivity index for input parameters of oral hazard model: **a** concentration of uranium (C_w), **b** exposure frequency (EF), **c** intake rate (IR), **d** body weight (BW) at different sample sizes considering two age-groups (children and adults)

case of adults for estimation of oral risk value (i.e., ingestion model). In terms of TOE, the highest Sobol score was recorded for $C_w = 77.46$, followed by IR = 20.12, EF = 9.87 and BW = 0.82 for children, whereas for adults, the highest Sobol score was found for $C_w = 85.58$, IR = 11.61 and EF = 10.65. These results indicated that intake rate of drinking water is most sensitive inputs among the two age-groups.

Figure 4 represents the Sobol scores for dermal model which consisted of six variables as model inputs. Results of ST showed that C_w, F and EF are the highly sensitive (> 0.1) parameters, whereas ET is sensitive and BW and SA are insensitive. Similar to ingestion model, the dermal model also showed higher value of Sobol scores for adults than for children. Interesting point to note here is that all the inputs converge and superimpose at higher sample sizes. Therefore, it is necessary to have a minimum of 100 samples while assessing the dose for U exposure via drinking water. Moreover, for children, BW also was relatively more sensitive as compared to adults in the case of dermal exposure. The optimal value of sample size was found as 12,000 samples for dermal model. For children, the value FOE was found as $C_w = 72.71$, followed by $F = 10.81$, EF = 8.43 and BW = 0.54, whereas for adults it was found as $C_w = 73.27$, $F = 10.9$, EF = 8.5, ET = 0.31 and BW = 0.02. From the results, it was found that surface area of the human body has no relative importance for dermal model (Fig. 6). For TOE, the highest Sobol score was recorded for $C_w = 80.18$, followed by $F = 15.03$ and EF = 11.85 for adults, whereas for children, the highest Sobol score was for $C_w = 79.74$, IR = 10.81 and EF = 8.43. The TOE was found to be higher for inputs C_w, EF and F for adults than the children.

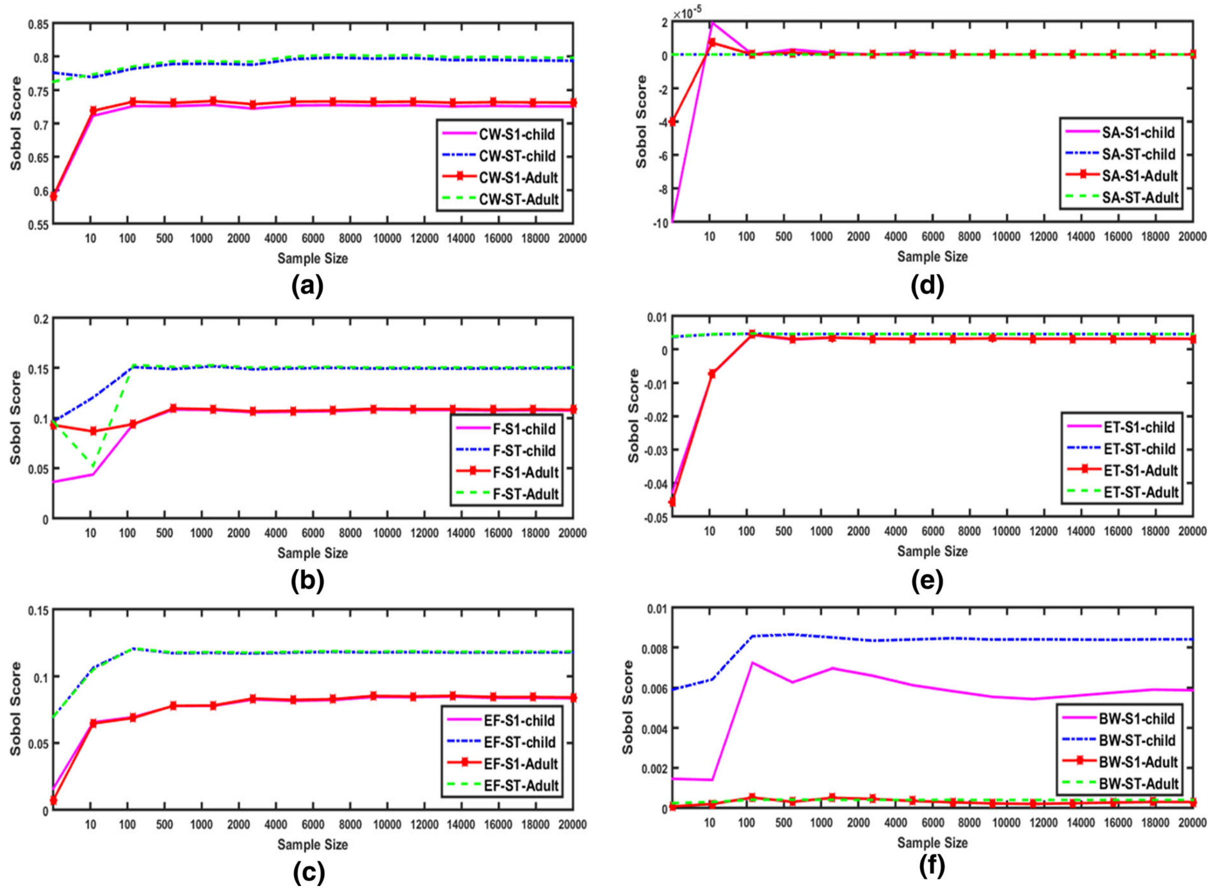

Fig. 4 Evolution of the first-order (S1) and total-order (ST) sensitivity index for input parameters of dermal hazard model: **a** concentration of uranium (C_w), **b** fraction of contact (F), **c** exposure frequency (EF), **d** surface area (SA), **e** exposure time (ET), **f** body weight (BW) at different sample sizes considering two age-groups (children and adults)

The second-order sensitivity index

The second-order sensitivity analysis explains the pairwise interactions among the inputs of the model. This also helps to analyze two-parameter interaction or pairwise variation to understand the most sensitive pairwise inputs. The second-order sensitivity index was performed for both HQ model to analyze the interaction effects of each variables.

For instance, the interactive Sobol indices (SOE) for ingestion model considering two age-groups were evaluated and are presented in Fig. 5. For children, the larger value of SOE was found for C_w _ IR, followed by C_w _ EF, C_w _ BW, IR _ EF, IR _ BW and EF _ BW. This result indicates that C_w _ IR are the significant input parameters for assessment of NCR rather than the C_w alone. The next highest value of interaction score was found for C_w _ EF. Moreover, larger value of interaction effects was observed for children for C_w _ IR, whereas in the case of adults, C_w _ EF indicated the larger value for interaction effects. These findings indicated that younger and older groups have synchronized effect with C_w, IR and EF, which was not possible to observe during the FOE calculation.

Figure 6 represents the Sobol indices of six variables for dermal model. Larger interaction effect (SOE) was observed for C_w _ $F = 3.6$, followed by C_w _ EF = 2.79 and F _ EF = 0.35 for adults. In the case of children, SOE was found highest for C_w _ $F = 3.54$, followed by C_w _ EF = 2.76 and F _ EF = 0.33. The decreasing order of interaction was found for C_w _ F, followed by C_w _ EF and F _ EF. This finding indicated that interaction effect of C_w and fraction of contact of skin (F) and exposure frequency (EF) are driving parameters for dermal risk as

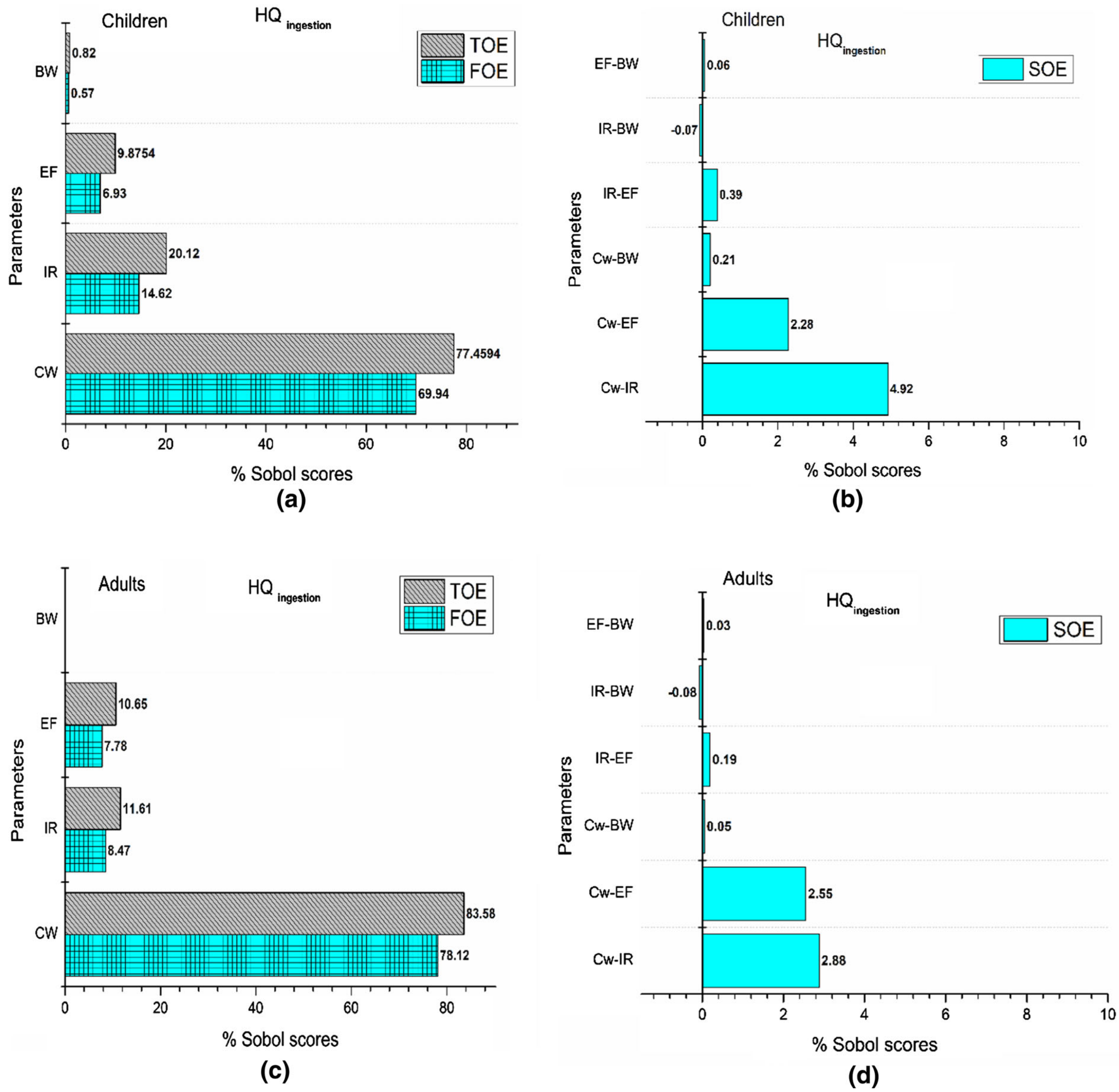

Fig. 5 Sensitivity analysis **a** first-order effect (FOE) and total effect (TOE) for children, **b** second-order effect (SOE) for children, **c** first-order effect (FOE) and total effect (TOE) for adults, **d** second-order effect (SOE) for adults considering HQ ingestion model

compared to the SA as negligible interaction effect was found between C_w and SA. Furthermore, it also indicates that the age-groups have no effect on dermal risk evaluation.

Conclusion

In this work, 156 representative samples of drinking water were investigated to determine the status of uranium concentration in the study region. Out of all the water samples, 17% of the samples exceeded the recommended level of U concentration in drinking water as per WHO guidelines. In addition, sensitivity and interaction effects of hazard model were investigated to understand the model behavior based on inputs. Sobol sensitivity analysis of FOE showed that the C_w parameter is the most influencing parameter followed by IR and EF, whereas BW has negligible effect for ingestion dose assessment. The Sobol value

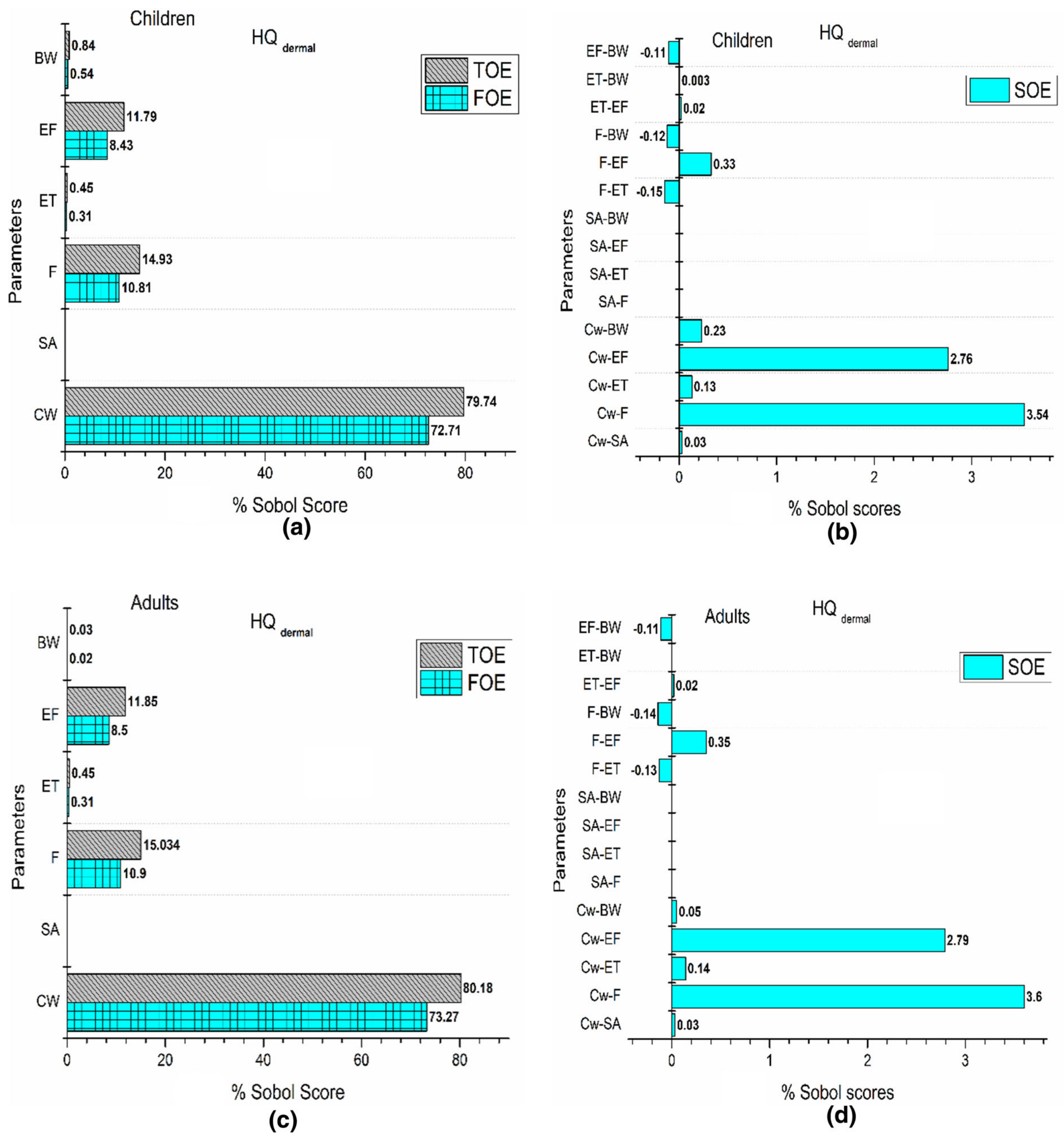

Fig. 6 Sensitivity analysis **a** first-order effect (FOE) and total effect (TOE) for children, **b** second-order effect (SOE) for children, **c** first-order effect (FOE) and total effect (TOE) for adults, **d** second-order effect (SOE) for adults considering HQ dermal model

of C_w and EF were found to be higher for adults, whereas IR was found higher for the children. This finding concluded that input parameter (IR) was more sensitive for younger age-groups than for the older groups. On the other hand, for the dermal risk model, SA showed no effect for both the age-groups (children and adults). The values of SOE indicate that interaction effect of C_w and IR (i.e., C_{w} _ IR) was the most sensitive parameter for both age-groups. Furthermore, the estimated value of SOE showed the C_w of U in groundwater and fraction of contact (F) to the skin were the most sensitive parameters for dermal model, which was not found during the FOE and TOE analysis. This result concluded that the older age-groups are more vulnerable than the younger age-groups during dermal exposure.

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Compliance with ethical standards

Conflict of interest The authors declare that they do not have any conflict of interest.

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