

# Estimation of distribution coefficient of uranium around a uranium mining site

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Received: 28 June 2012 / Published online: 2 September 2012  
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**Abstract** Distribution coefficient ( $K_d$ ) of uranium and its daughter products are very important for migration study around uranium mining sites. Since the distribution coefficient depends very much on the soil and groundwater chemistry, generation of site specific  $K_d$  is very important. In the literature there is a large variation of  $K_d$  values of uranium. For realistic prediction of contaminant migration, literature  $K_d$  value is not very effective. So site specific experimental  $K_d$  values are required. The present study emphasizes on the estimation of site specific distribution coefficient for uranium around a uranium mining site. The soil and groundwater parameters which affect the  $K_d$  value of uranium have also been estimated. Soil and groundwater samples from nine locations around Turamdih uranium mining site were collected and chemically characterized for various parameters. The distribution coefficient of uranium in top and one meter depth soil samples from above locations were estimated using laboratory batch method. The distribution coefficient of uranium varies from  $69 \pm 4$  to  $5524 \pm 285$  l/kg. No significant difference in uranium  $K_d$  values was observed for top and one meter depth soil samples. In the top and one meter depth soil samples uranium  $K_d$  values vary from  $129 \pm 8$  to  $5524 \pm 285$  and  $69 \pm 4$  to  $3862 \pm 195$  l/kg respectively. For the estimation of distribution coefficient of uranium different parameters like equilibration time, solid to solution ratio, method of tracer addition to solution, solid-solution separation method etc. have been optimized. The distribution coefficient of uranium determined in the present study will

be used for the migration study of uranium around uranium mining sites.

**Keywords** Naturally occurring radionuclides · Uranium mining · Physico chemical characterization · Distribution coefficient · Laboratory batch method

## Introduction

Naturally occurring radionuclides are present in many natural resources. High concentration of these radionuclides are often found in certain geological materials, namely igneous rocks and ores. Activities involving the extraction, exploitation and processing of materials containing NORM include the mining and processing of uranium and metal ores, the combustion of fossil fuels, production of natural gas and oil, and the phosphate industry. NORM-containing wastes from outside the nuclear fuel cycle have received increasing attention lately, due not only to the existing large amounts, but to the potential long-term hazards resulting from the fact that NORM comprise long-lived radionuclides with relatively high radiotoxicities [1]. The mobilization of U away from the primary ore body occurred when the weathering front, which is slowly moving downwards, intersected the primary U mineralisation. This event occurred between 1 and 3 million years ago [2]. Groundwater then oxidised and dissolved uraninite and other minerals from the primary ore and transported the U(VI) in the groundwater in the form of uranyl-carbonate complexes such as  $UO_2CO_3$  and  $UO_2(CO_3)_2^{2-}$  [3]. From uranium mining there is a chance of contamination by trace metals and naturally occurring radionuclides. Any metals or other contaminants that are present in the uranium tailings may be leached into the soil and enter into the groundwater. The possible pathway of the

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metals to reach the environment is via the leaching of the metals from tailings to the aquifer below and subsequent transport in the aquifer. Migration of metals and other tailings constituents occurs through leaching and erosion. The degree of migration is related to numerous factors including the chemistry of the tailings material, the permeability of the impoundment and liner (if present), the amount of precipitation, the nature of the underlying soils and the proximity to both surface and groundwater. Predicting the transfer of radionuclides in the environment for normal release, accidental, disposal or remediation scenarios in order to assess exposure requires the availability of an important number of generic parameter values. One of the key parameters in environmental assessment is the solid liquid distribution coefficient,  $K_d$ , which is used to predict radionuclide–soil interaction and subsequent radionuclide transport in the soil column [4]. The radionuclides present at these sites can enter the food chain directly via the soil–plant–animal pathway, or indirectly by the use of contaminated groundwater or surface water for irrigation purposes or drinking water. To assess the uptake in the food chain and by biota in general and to predict human exposure, knowledge on the environmental parameters governing radionuclide mobility and uptake is indispensable. Distribution coefficient values of uranium are very much dependent on various physico chemical parameters of soil and groundwater, that's why site specific distribution coefficient is very much important for the prediction of the contaminant transport in geological matrices. Uranium sorption is affected by soil properties other than soil texture such as pH,  $\text{CaCO}_3$ , content of amorphous iron oxides, soil organic matter content, cation exchange capacity, and phosphate status [5]. Adsorption of U(VI) on Fe-oxides and clays appears to play a major role in reducing U mobility [6]. Soil solution distribution coefficient,  $K_d$ , that describe the mobility of elements in soils and rocks range in different environments from 0.03 in sandy soils to 20,000 in clayey soils [7–9]. Soil variability within a specific area is sometimes large enough to have  $K_d$  values ranging over five orders of magnitude [8]. The uncertainty associated with  $K_d$  values for several radionuclides results in a large variability of the predicted dose for humans after calculations, even when considering only site-specific soils. To understand the variability of uranium  $K_d$  in soil, it is necessary to focus on the parameters influencing the mobility of uranium in soils.

In the present study the soil from top and one meter depth and groundwater samples collected from nine different locations around Turamdih uranium mining site were chemically characterized and distribution coefficient for uranium was estimated in soil samples by using laboratory batch method. Since distribution coefficient of uranium is a strong function of soil and ground water parameters, the effect of these parameters on the  $K_d$  values of uranium have been found out.

## Materials and methods

### Sampling location

Soil and groundwater samples were collected from nine different places, around Turamdih uranium mining area, Jharkhand, India (near Turamdih Tailings Pond (TP), Bada Talsa, Dhatkidih, Keruadungri, Kachha, Ghagidih Exit, Jagannathpur, Gilinguda and Giridih). In India the first uranium mining and ore processing operations commenced in the mid-1960s at Jaduguda in the eastern part of the country. The position of the central part of Jaduguda is  $\text{E}86^\circ20'$ ;  $\text{N}22^\circ40'$ . Jharkhand state has a long slender area called “East Singhbhum” which lies between West Bengal and Orissa. Other three uranium ore deposits at Turamdih, Narwapahar, and Bhatin, are all located in this area and taken up for underground mining. The position of the central part of Turamdih is  $\text{E}86^\circ11'$ ;  $\text{N}22^\circ43'$ .

### Sample collection and processing

All the soil samples were collected with a wooden spatula below 10 cm and 1 m depth and kept in polyethylene bags previously treated with one molar solution of hydrochloric acid and rinsed with distilled water. The dry soil samples ( $<2$  mm) were homogenized and used for chemical characterization as well as estimation of distribution coefficient of uranium. Same soil samples ( $<2$  mm) were used for estimation of pH,  $\text{CaCO}_3$ , cation exchange capacity (CEC), organic matter and organic carbon. Groundwater samples were collected in plastic carboy previously treated with one molar solution of hydrochloric acid and rinsed with distilled water. Groundwater samples were filtered through  $0.45 \mu$  filter paper and were used for the estimation of anions ( $\text{SO}_4^{2-}$ ,  $\text{NO}_3^-$ ,  $\text{F}^-$ ,  $\text{Cl}^-$  and  $\text{HCO}_3^-$ ), inorganic carbon and organic carbon. Filtered groundwater samples were digested by using electronic grade nitric acid and metals were analysed in the digested water samples. Soil samples ( $<2$  mm) were microwave digested and metals were analysed in the digested soil samples. All chemicals used were Merck, Suprapur, Analar or electronic grade.

## Methodology

### Soil characterization

#### *Particle size distribution in soil*

Soil samples were air dried and sieved using different mesh sieves (2 mm, 300 and  $45 \mu\text{m}$ ) in a electromagnetic sieve shaker (EMS 08) to find out the particle size distribution in order to know the soil type.

### Estimation of pH in soil

The pH of soil was estimated using the methodology IS 2720, Part No. 26. For that 30 g soil (<2 mm) was taken in a 100 ml beaker and 75 ml of double distilled water was added to it. The suspension was stirred for a few seconds and covered with a watch glass and allowed to stand for 1 h with occasional stirring. The soil solution pH was estimated using a pH meter (DPH-500) which was calibrated using standard buffer (pH 4, 7 and 9) before testing.

### Estimation of calcium carbonate in soil

Calcium carbonate was estimated by using the methodology IS 2720, Part No. 23. 5 g of soil (<2 mm) was taken in a 150 ml beaker and 100 ml of 0.1 (N) HCl was added to it. The solution mixture was covered with watch glass and vigorously stirred occasionally for 1 h. After settling 20 ml of the supernatant liquid was taken in a conical flask and 6–8 drops of bromothymol blue indicator was added to it and titrated against 0.1 (N) NaOH solution.

### Estimation of cation exchange capacity (CEC) in soil

The values of CEC were obtained by calculation from exchangeable cations. 1 g of soil was taken in a 100 ml beaker, 18 ml CH<sub>3</sub>COONH<sub>4</sub> (1 N) was added, kept for 1 h under stirring condition. Allowed to settle and filtered, remaining soil was washed three times with 1 (N) CH<sub>3</sub>COONH<sub>4</sub>. Again soil was washed with absolute alcohol (10 ml). Evaporated and final volume was made up to 10 ml. Na<sup>+</sup>, K<sup>+</sup>, Ca<sup>++</sup> and Mg<sup>++</sup> was analysed in this sample by using atomic absorption spectrometry (AAS) (GBC Avanta). Sum of the equivalent concentration of these ions give CEC in soil.

### Carbon content in soil

Organic carbon was analysed in soil samples (<2 mm) using carbon analyzer (Shimadzu TOC).

### Estimation of organic matter content in soil

Organic matter in the samples were estimated by weight loss on ignition method. 10 g of dry soil (105 °C) sample (<2 mm) was taken in a porcelain crucible covered with watch glass. The soil was heated in a muffle furnace at 360 °C [10, 11] for 2 h and after that final weight was taken. Then loss on ignition (LOI) was calculated by using the following equation

$$\text{LOI (\%)} = \frac{(\text{weight at } 105\text{ }^{\circ}\text{C} - \text{weight at } 360\text{ }^{\circ}\text{C}) \times 100}{\text{weight at } 105\text{ }^{\circ}\text{C}}$$

Ben-Dor and Banin [12] showed that ignition temperatures in the range of 400–430 °C resulted in no significant bias from thermal breakdown of carbonates. Jackson [13] and Ball [14] both recommended ignition temperatures of <400 °C to minimize weight loss from carbonates, structural waters in clays and hydrated salts.

### Groundwater characterization

In filtered water samples anions (Cl<sup>-</sup>, F<sup>-</sup>, NO<sub>3</sub><sup>-</sup>, SO<sub>4</sub><sup>-</sup>) were analysed using ion chromatograph (IC Metrohm 733) and HCO<sub>3</sub><sup>-</sup> was analysed by titration. Organic and inorganic carbon were analysed using carbon analyzer (Shimadzu TOC). pH was estimated using pH meter (DPH-500) which was calibrated using standard buffer (pH 4, 7 and 9) before testing.

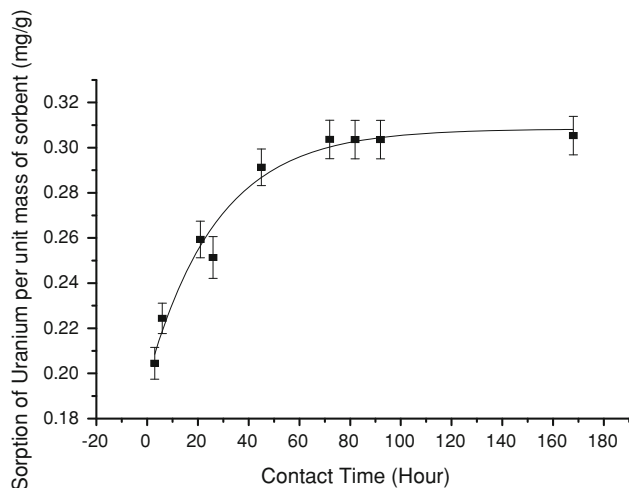
### Optimization of different parameters for the estimation of distribution coefficient of uranium by using laboratory batch method

#### Sorption experiment for determination of equilibration time

The sorption experiment has been carried out by laboratory batch method [15, 16]. Natural uranium was used as a tracer. In 50-ml polypropylene centrifuge tubes soil (1 g) and natural uranium (400 µg) in 30 ml groundwater (1:30 S/L ratio) were shaken at room temperature. The experiment was carried out in different time interval (0–170 h) to find out the equilibration time shown in Fig. 1, which clearly indicates that sorption equilibrium for uranium achieved at around 72 h. The laboratory batch method was carried out in a dual rotating shaker bath Model (SK-300) with a shaking rate of 0.8–1.2 oscillations per second. Solution mixture was centrifuged using high speed centrifuge (6,000 rpm) R-8C-DX and then filtered through 0.45 µm filter paper. Uranium was analysed in the filtrate using differential pulse adsorptive stripping voltammeter (DPASV) (663 VA Stand Metrohm). The amount of adsorbed uranium was estimated from the difference of the uranium concentrations in the aqueous phase before and after the adsorption.

#### Method of tracer addition

Addition of tracer to a solution represents a critical step in the execution of radionuclide migration studies. Two items must be carefully considered: (1) the total amount of tracer added must be soluble in the volume of solution used and (2) the chemical composition of the groundwater or



**Fig. 1** Determination of equilibration time for sorption of uranium in soil. Soil (<2 mm) and groundwater from the same location Gilinguda, initial uranium concentration 400  $\mu\text{g}$ , room temperature, soil to water ratio 1:30)

synthetic groundwater must remain unchanged, except for the addition of the radionuclide(s) to be studied. In the present experiment 400  $\mu\text{g}$  of U(VI) in 5 %  $\text{HNO}_3$  was added with the soil solution mixture. After addition of tracer pH of the solution was adjusted up to the same level when the soil solution equilibrium was there prior to tracer addition by using 10 (M) NaOH solution.

#### Analysis of uranium

The principle of the determination is to pre concentrate uranium(VI) from the sample solution as the chloranilic acid (CAA) complex, potentiostatically on a hanging mercury drop electrode (HMDE) and subsequently to dissolve it again by applying a cathodic potential scan. A peak occurs in the voltammogram as a result of the reduction of uranium(VI) which can be evaluated quantitatively [17].

0.1 ml digested sample, 9.4 ml double distilled water, 50  $\mu\text{l}$  of 0.1 (M) EDTA (ethylene diamine tetraacetic acid) and 400  $\mu\text{l}$  CAA were pipetted into cell. pH of the resulting

solution was adjusted to 1.9 using  $\text{HNO}_3/\text{NaOH}$  and de aerated using IOLAR-II grade nitrogen. An accumulation potential of 0.03 V is applied to a fresh mercury drop electrode, while the solution is stirred for an accumulation time of 60 s at 2,000 rpm. Following the pre concentration, stirring is stopped and after equilibrium time of 5 s, the differential pulse voltammograms are recorded from 0.03 to  $-0.11$  V, at a scan rate of  $10 \text{ mV s}^{-1}$  and pulse amplitude of 50 mV. All samples are analysed by standard addition method with  $1 \mu\text{g ml}^{-1}$  uranium. Minimum detectable activity found for this method was  $0.08 \text{ ng ml}^{-1}$ .

Calculation for estimation of distribution coefficient of uranium in soil

Distribution coefficient is defined as the ratio of the concentration of the uranium in the sorbent per unit mass and in the solution per unit volume at equilibrium.

The distribution coefficient was calculated as per the following equation:

$$K_d = \left( \frac{C_0 - C_e}{C_e} \right) \times \frac{V}{m}$$

where  $C_0$  is the concentration of the initial solution ( $\mu\text{g/ml}$ ),  $C_e$  is the concentration of the solution in equilibrium ( $\mu\text{g/ml}$ ),  $V$  is the volume of the solution (ml),  $m$  is the amount of the adsorbent (g).

## Results and discussion

Chemical characterization of groundwater and soil samples

pH, concentration of inorganic carbon, organic carbon, bicarbonate, fluoride, chloride, nitrate, sulphate and uranium concentration in the groundwater samples collected around uranium mining area are shown in the Table 1. The values were also compared with other reported values [18, 19].

**Table 1** Mean and ranges for the descriptive parameters of the groundwater samples collected from nine different locations around Turamdih uranium mining area

Parameters	Mean	Range	Reported value	Reference
pH	7.02	6.82–7.27	8.0–9.40	Acharya et al. [18]
Inorganic carbon (mg/l)	37.51	18.36–57.2	–	–
Organic carbon (mg/l)	11.81	3.2–22.1	–	–
Bicarbonate ( $\text{HCO}_3^-$ ) (mg/l)	153.5	80.7–214.4	176.9–915	Acharya et al. [18]
Fluoride ( $\text{F}^-$ ) (mg/l)	0.813	0.26–1.73	0.20–1.10	Acharya et al. [18]
Chloride ( $\text{Cl}^-$ ) (mg/l)	88.4	3.47–442.3	35.5–692.25	Acharya et al. [18]
Nitrate ( $\text{NO}_3^-$ ) (mg/l)	32.6	0.29–197.6	–	–
Sulphate ( $\text{SO}_4^-$ ) (mg/l)	55.5	10.4–232.7	0–144	Acharya et al. [18]
Uranium concentration (ng/ml)	5.43	1.05–10.25	1.39–98.25	Rani et al. [19]

The particle size distribution of soil samples has been given in Table 2. Which shows that the soil samples are mostly sandy type. pH, concentration of calcium carbonate, cation exchange capacity (CEC), organic matter, organic carbon and uranium concentration for soil samples are shown in Table 3. The values were also compared with other reported values [20–22].

Estimation of distribution coefficient of uranium around Turamdih uranium mining area

To carry out the batch experiments natural uranium was used as a tracer. The distribution coefficients ( $K_d$ ) for uranium at different places around Turamdih uranium mining site were determined shown in Fig. 2. The  $K_d$  for uranium varies from  $69 \pm 4$  to  $5524 \pm 285$  l/kg. In the top and one meter depth soil samples uranium  $K_d$  values vary from  $129 \pm 8$  to  $5524 \pm 285$  and  $69 \pm 4$  to  $3862 \pm 195$  l/kg respectively. The  $K_d$  values obtained from the present study ( $69 \pm 4$ – $5524 \pm 285$  l/kg) are compared with the reported values [7] for all type of soils. Sheppard et al. [23] compiled the  $K_d(U)$  values for soils within a pH range of 4–8.8, and they recommended  $K_d(U)$  values of 40, 200, 200 and 2,000 l/kg for sand, loam, clay, and organic soils respectively.

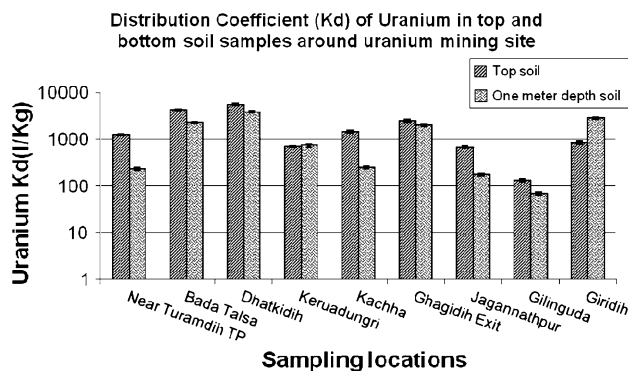


Fig. 2 Distribution coefficient of uranium in soil around Turamdih uranium mining area

Soil and groundwater parameters affecting distribution coefficient of uranium

Distribution coefficient of uranium is very much dependent on different soil and groundwater parameters. That’s why estimation of site specific distribution coefficient is very much important for migration study.

Effect of soil pH on uranium  $K_d$

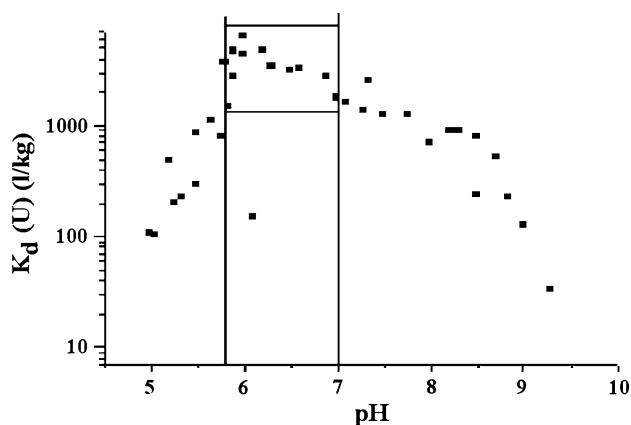
In the present study it is observed that with increasing soil pH from 5 to 6  $K_d(U)$  increases linearly ( $R^2 = 0.87$ ). But reverse scenario occurs when soil pH exceeds 6. In that case  $K_d(U)$  decreases linearly ( $R^2 = 0.469$ ) shown in the Fig. 3. Actually at the pH range 5.8–7.0 highest  $K_d$  values of uranium is observed. Uranium can exist in the +3, +4, +5, and +6, oxidation states in aqueous environments. Dissolved U(III) easily oxidizes to U(IV) under most reducing conditions found in nature. The U(V) aqueous species readily disproportionates to U(IV) and U(VI). Consequently, U(IV) and U(VI) are the most common oxidation states of uranium in nature. Uranium will exist in the +6 and +4 oxidation states, respectively, in oxidizing and more reducing environments. At pH below 5, U(VI) is present as the uranyl ion ( $UO_2^{2+}$ ). At a higher pH, the uranyl ion hydrolyzes, forming a number of aqueous hydroxide complexes, which dominate U(VI) speciation in the absence of dissolved inorganic ligands (carbonate,

Table 2 Particle size distribution of soil samples

Sampling locations	Between 300 $\mu$ m and 2 mm (%)	Between 45 and 300 $\mu$ m (%)	<45 $\mu$ m (%)
Near Turamdih TP	45.04	44.42	10.54
Bada Talsa	46.31	41.85	11.84
Dhatkidih	75.8	20.61	3.59
Keruadungri	50.74	33.82	15.44
Kachha	55.34	36.26	8.40
Ghagidih Exit	40.48	54.03	5.48
Jagannathpur	41.64	52.66	5.70
Gilinguda	31.45	57.9	10.65
Giridih	53.53	40.19	6.28

Table 3 Mean and ranges for the descriptive parameters of the soil samples collected from nine different locations around Turamdih uranium mining area

Parameters	Mean	Range	Reported value	Reference
pH	6.75	5.0–8.9	7.2–10.0	Nuria et al. [20]
CaCO <sub>3</sub> (%)	0.53	0.20–1.49	1.2–57.9	Nuria et al. [20]
CEC (meq/100 g)	9.01	2.86–20.15	2.0–9.0	Hepper et al. [21]
Organic matter (%)	5.25	3.5–7.47	–	–
Organic carbon (%)	0.44	0.31–0.60	0.1–2.2	Nuria et al. [20]
Uranium concentration ( $\mu$ g/g)	2.13	1.32–2.67	0.75–2.06	Sharma et al. [22]



**Fig. 3** Variation of distribution coefficient of uranium with pH in soil samples around Turamdih uranium mining area

fluoride, sulphate and phosphate). At the pH range of 6–10, highly soluble carbonate complexes dominate (the acid biphosphate, bicarbonate and tricarbonato) [24]. Aqueous pH is likely to have a profound effect on U(VI) sorption to solids. There are two processes by which it influences sorption. First, it has a great impact on uranium speciation such that poorer-adsorbing uranium species will likely exist at pH values between about 6.5 and 10. Secondly, decreases in pH reduce the number of exchange sites on variable charged surfaces, such as iron-oxides, aluminum-oxides, and natural organic matter. Echevarria et al. [25] they found a significant relation between soil  $K_d$  and pH. For soils in 5.5–8.8 pH range they deduced a linear relationship:  $\log K_d = -1.29 (\pm 0.17) \times \text{pH} + 11.0 (\pm 1.2)$ ,  $R^2 = 0.76$ . Vandenhove et al. [26] examined the effect of soil pH on uranium availability for 18 spiked soils, and a similar linear decrease of  $\log K_d$  with pH was observed for soils with  $\text{pH} \geq 6$  [ $\log K_d = -1.18 \times \text{pH} + 10.8$ ,  $R^2 = 0.65$ ], which was explained by the increased amount of soluble uranyl–carbonate complexes at high pH. In general, the adsorption of uranium by soils and single-mineral phases in carbonate-containing aqueous solutions is low at pH values less than 3, increases rapidly with increasing pH from 3 to 5, reaches a maximum in adsorption in the pH range from 5 to 8, and then decreases with increasing pH at pH values greater than 8. This trend is similar to the in situ  $K_d$  values reported by Serkiz and Johnson [27], and percent adsorption values measured for uranium on single mineral phases such as those reported for iron oxides [28–31], clays [30, 32, 33], and quartz [30]. This pH-dependent behavior is related to the pH-dependent surface charge properties of the soil minerals and complex aqueous speciation of dissolved U(VI), especially near and above neutral pH conditions where dissolved U(VI) forms strong anionic uranyl–carbonato complexes with dissolved

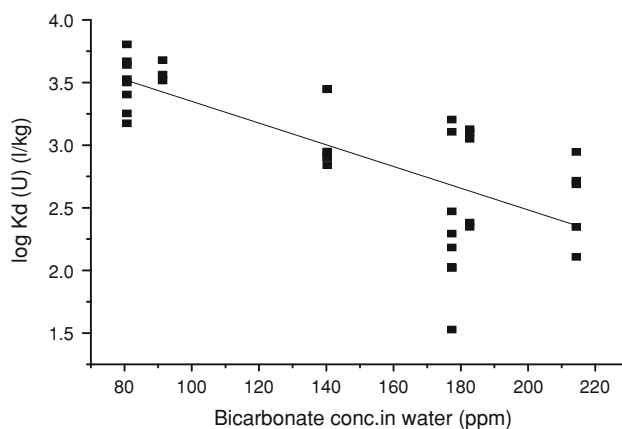
carbonate. The pH of the solution also plays an important role in the adsorption of radionuclides on the adsorbents, and influences the metal speciation and surface binding sites. The effect of pH on uranium sorption was investigated with pH ranging from 2 to 10, while keeping all other parameters constant as described previously [34].

#### Effect of bicarbonate in groundwater on uranium $K_d$

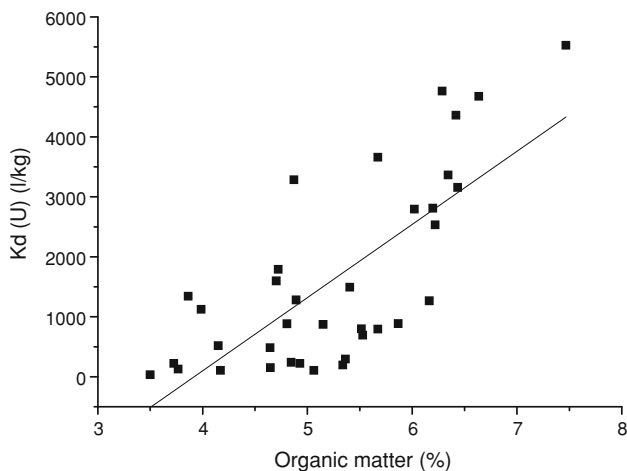
In the soil samples collected around uranium mining site, it is observed that with increasing concentration of bicarbonate in groundwater, distribution coefficient value of uranium decreases ( $R^2 = 0.52$ ) due to formation of soluble uranyl–carbonate complex, shown in the Fig. 4. In presence of  $\text{HCO}_3^-$ , U(VI) forms dissolved uranyl–carbonate complex, so adsorption of uranium in solid matrix decreases. According to McKinley et al. [32] dissolved carbonate has a significant effect on the aqueous chemistry and solubility of dissolved U(VI) occurs through the formation of strong anionic carbonato complexes. In turn, this complexation affects the adsorption behavior of U(VI) at alkaline pH conditions.

#### Effect of organic matter and organic carbon on uranium $K_d$

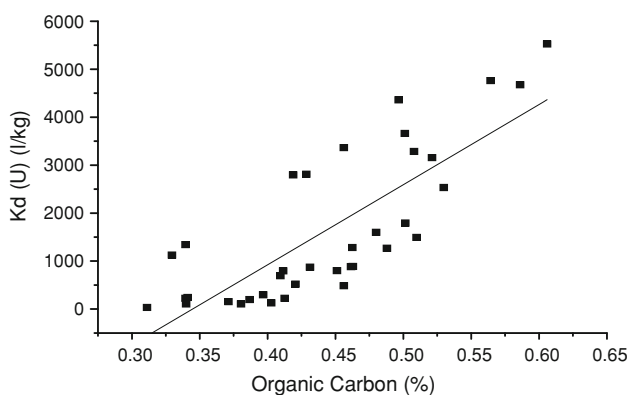
In the present study with increasing concentration of organic matter and organic carbon in the soil samples distribution coefficient of uranium increases linearly ( $R^2 = 0.54, 0.61$ ), shown in Figs. 5 and 6. Organic matter and clay minerals provide exchange sites and as such are expected to increase sorption of  $\text{UO}_2^{2+}$  and other positively-valent U-forms. The influence of organic matter (OM) on U mobility is twofold: a decreased mobility through sorption by exchange and an increased mobility following formation of organic complexes and colloids.



**Fig. 4** Variation of distribution coefficient of uranium with  $\text{HCO}_3^-$  in ground water samples around Turamdih uranium mining area



**Fig. 5** Variation of distribution coefficient of uranium with organic matter content in soil samples around Turamdih uranium mining area



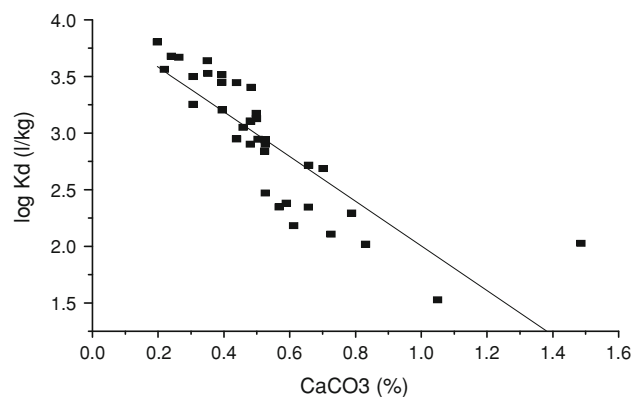
**Fig. 6** Variation of distribution coefficient of uranium with organic carbon content in soil samples around Turamdih uranium mining area

In absence of large amounts of organic matter and clay, U is considered to be relatively mobile. For soils with  $\text{pH} < 6$ , Vandenhove et al. [26] reported very significant correlations between  $K_d(\text{U})$  and organic matter [ $K_d(\text{U}) = 1963 \times \text{OM} - 5432$ ,  $R^2 = 0.78$ ].

#### Effect of $\text{CaCO}_3$ on uranium $K_d$

In the present study with increasing concentration of  $\text{CaCO}_3$  in the soil samples the distribution coefficient value of uranium decreases linearly ( $R^2 = 0.69$ ), shown in Fig. 7. This type of variation of distribution coefficient of uranium with respect to calcium carbonate in soil is observed may be due to with increasing calcium carbonate concentration pH of soil decreases due to formation of bicarbonic acid and  $K_d(\text{U})$  decreases accordingly.

The effect of cation exchange capacity on distribution coefficient of uranium is not found out in our present study.



**Fig. 7** Variation of distribution coefficient of uranium with calcium carbonate content in soil samples around Turamdih uranium mining area

## Conclusions

The distribution coefficient of uranium varies from  $69 \pm 4$  to  $5524 \pm 285$  l/kg in soil samples collected from Turamdih uranium mining site. No significant difference in uranium  $K_d$  values was observed for top and one meter depth soil samples. In the present study it is clearly observed that distribution coefficient of uranium is very much dependent on different soil parameters like pH, organic carbon, organic matter, calcium carbonate and groundwater parameter like bicarbonate concentration. With increasing soil pH from 5 to 6  $K_d(\text{U})$  increases linearly ( $R^2 = 0.87$ ). But reverse scenario occurs when soil pH exceeds 6. In that case  $K_d(\text{U})$  decreases linearly ( $R^2 = 0.469$ ). With increasing concentration of bicarbonate in groundwater, distribution coefficient value of uranium decreases ( $R^2 = 0.52$ ) due to formation of soluble uranyl-carbonate complex. With increasing concentration of organic matter and organic carbon in the soil samples distribution coefficient of uranium increases linearly ( $R^2 = 0.54, 0.61$ ). It is observed that with increasing concentration of  $\text{CaCO}_3$  in the soil samples, the distribution coefficient value of uranium decreases linearly ( $R^2 = 0.69$ ). Estimated site specific distribution coefficient values of uranium will be useful for prediction of contaminant transport from a uranium mining site.

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