

A Review of the Distribution Coefficient (K_d) of Some Selected Heavy Metals over the Last Decade (2012-2021)

Chabi Dari Seidou¹, Tiejun Wang^{1,2,3*}, Mikouendanandi Mouendo Rahmat Brice Espoire¹, Yibin Dai¹, Yutao Zuo¹

¹Institute of Surface-Earth System Science, School of Earth System Science, Tianjin University, Tianjin, China

²Tianjin Bohai Rim Coastal Earth Critical Zone National Observation and Research Station, Tianjin University, Tianjin, China

³Tianjin Key Laboratory of Earth Critical Zone Science and Sustainable Development in Bohai Rim, Tianjin University, Tianjin, China

Email: *tiejun.wang@tju.edu.cn

How to cite this paper: Seidou, C. D., Wang, T. J., Espoire, M. M. R. B., Dai, Y. B., & Zuo, Y. T. (2022). A Review of the Distribution Coefficient (K_d) of Some Selected Heavy Metals over the Last Decade (2012-2021). *Journal of Geoscience and Environment Protection*, 10, 199-242.

<https://doi.org/10.4236/gep.2022.108014>

Received: July 22, 2022

Accepted: August 27, 2022

Published: August 30, 2022

Copyright © 2022 by author(s) and Scientific Research Publishing Inc.

This work is licensed under the Creative Commons Attribution International License (CC BY 4.0).

<http://creativecommons.org/licenses/by/4.0/>



Open Access

Abstract

This review synthesizes the methods for estimating the distribution coefficient (K_d) and provides a compilation of K_d values for five heavy metals (As, Pb, Cd, Cu, and Zn) based on research published in the last decade (2012-2021). We used the PRISMA method to ensure the transparency of the collected data. For mono-metal systems (MS), the K_d values ranged from 10^{-2} to 10^7 L/kg for Pb, from 10^{-2} to 10^6 L/kg for Cd, As, and Zn, and from 10^{-2} to 10^5 L/kg for Cu. In competitive systems (CS), the K_d values ranged from 10^{-2} to 10^5 L/kg for Cu, and 10^{-2} to 10^4 L/kg for Pb, Cd, and Zn, while no K_d value for As was reported under CS. It was found that the K_d values of heavy metals are affected not only by soil chemical and physical properties but also by the nature and characteristics of the metal involved along with experimental conditions. The total references number of K_d data observation per element metal are represented as follows: Cd 35 (50%), Zn 35 (50%), Pb 33 (47.14%), Cu 33 (47.14%), and As 19 (27.14%). Overall, most research was done 1) on MS rather than CS, 2) on sorption rather than desorption, 3) on soil rather than sediments, and 4) most literature have reported the K_d values, derived from batch method than on column method. Despite significant progress over the past decade towards a better understanding of the variation in K_d values and the effect of factors influencing them to provide important parameters for predicting and controlling toxic metals in soils, additional research is still warranted to the complexity of underlying processes.

Keywords

K_d Values, Heavy Metals, PRISMA, Sorption, Desorption

1. Introduction

The presence of heavy metals in soil and groundwater has long been recognized as a contentious issue worldwide. Because of their inherent accumulative and nondegradable properties, heavy metals in soil and groundwater pose significant environmental risks (Guo et al., 2018; Vatandoost et al., 2018; Yang et al., 2021). The availability of heavy metals to plants and the risk of these metals entering groundwater depend on their sorption and desorption from soils. Therefore, to assess their environmental risks and develop appropriate remediation strategies, estimation of sorption and adsorption capacities of heavy metals is required. The heavy metal sorption capacity, also known as distribution coefficient (K_d), is determined by the ratio of heavy metal concentrations in the solid phase to that in equilibrium solution after a given reaction time. When measured under the same experimental conditions, K_d is a valuable parameter for comparing the sorptive capacities of different soils (Ding et al., 2018; Kumar et al., 2019; Shaheen et al., 2013) and essential for simulating the transport of heavy metals into environments.

Studies have implemented several laboratory approaches to determining K_d values of heavy metals. The factors influencing K_d values include sorption systems (either mono-metal system-MS or competitive system-CS), element properties (e.g., element type and characteristics), physical and chemical properties of soil (e.g., soil texture, mineralogical composition, and cation exchange capacity, and soil pH), and experimental conditions (e.g., reaction time and temperature) (Braz et al., 2013; Gu et al., 2014; Guo et al., 2013; Kumar et al., 2019; Loganathan et al., 2012; Wang et al., 2022; Zheng et al., 2016). The diversity of influencing factors eventually results in a wide range of K_d values for heavy metals in soils, which can make it difficult to deduce some standard K_d values or magnitudes to be used as reliable indicators for assessing the remediation process of toxic metals in soils. As a result, these differences in K_d must be accounted for when predicting and managing heavy metal contamination of soils, sediment, and groundwater. A constant update of recently established K_d values of heavy metals may be required for a thorough understanding of the influence of various factors on heavy metal K_d . However, there is a need to have a review to update the recent progress of different studies on K_d of heavy metals in soil. Several review articles have already been published on the significance and role of influencing factors of K_d values (Pathak et al., 2014), and the role of heavy metal K_d in mobility assessment (Shaheen et al., 2013). Numerous individual research articles have recently provided new values of K_d and information on various aspects of this topic. In addition, arsenic (As), lead (Pb), cadmium (Cd), copper

(Cu), and zinc (Zn) are among the most commonly found heavy metals in the environment, causing human health and environmental risks (Jaishankar et al., 2014).

Therefore, the current work aims to synthesize the existing methods for estimating K_d values and provide compilations of K_d data of five heavy metals, including As, Pb, Cd, Cu, and Zn, based on the recent research progress between 2012 and 2021.

2. Experimental Method of K_d Estimation

At the equilibrium state, the metal K_d is defined as (Allison & Allison, 2005):

$$A = C_i + A_i \quad (1)$$

$$K_d = \frac{\text{Mass of Adsorbete Sorbed}}{\text{Mass of Adsorbete in Solution}} = \frac{\text{Sorbed Metal Concentration (mg/kg)}}{\text{Dissolved Metal concentration (mg/L)}} = \frac{A_i}{C_i} \quad (2)$$

where A ($\frac{\text{mg}}{\text{kg}}$ or $\frac{\text{mg}}{\text{L}}$), C_i ($\frac{\text{mg}}{\text{L}}$), and A_i ($\frac{\text{mg}}{\text{kg}}$) are free or unoccupied surface adsorption sites, total dissolved adsorbate remaining in solution at equilibrium, and the amount of adsorbate on the solid at equilibrium respectively.

K_d values are measured using one of the five general methods: the laboratory batch method, the laboratory flow-through (or column) method, the in-situ batch method, the field modeling method, and the K_{oc} method (EPA, 2004; Kumar et al., 2019). The following section briefly summarizes the advantages and disadvantages of each method.

2.1. Laboratory Batch Method

The laboratory method is commonly used to determine K_d values in batch studies. This method involves spiking a solution with heavy metals, mixing the spiked solution with a solid for a certain amount of time, separating the solution from the solid, and measuring the concentration of the spiked heavy metal remaining in the solution (Allison & Allison, 2005; EPA, 2004). Equation (3) is then used to calculate the concentration of adsorbate sorbed on the solid phase (A_s , also known as q_i).

$$A_i = q_i = \frac{V_w (C_0 - C_i)}{M_{sed}} \quad (3)$$

Substitution of Equation (3) into Equation (2) gives:

$$K_d = \frac{V_w (C_0 - C_i)}{M_{sed} C_i} \quad (4)$$

The batch method has the primary advantage of allowing such experiments to be completed quickly for various range of elements and chemical environments. The major disadvantage of the batch method for measuring K_d is that it fre-

quently fails to replicate the exact conditions of chemical reactions.

2.2. Laboratory Flow-Through (or Column) Method

The flow-through experiments are the second most commonly used method for measuring K_d values. It provides a more realistic simulation of field conditions and quantifies the movement of contaminants relative to groundwater flow. The basic experiment is completed by passing a liquid laced with the desired known amount of contaminant and nonadsorbing tracer solutions through a soil column of packed soil with known bulk density and porosity. The resulting data are represented graphically as a breakthrough curve. The retardation factor (R_f) is defined as the ratio of pore-water velocity (V_p , cm/hr) to contaminant velocity (V_c , cm/hr), Equation (5), and it is frequently calculated directly from experimental data.

$$R_f = \frac{V_p}{V_c} \quad (5)$$

The pore-water velocity is operationally defined as the velocity of the nonadsorbing tracer.

The following equations can calculate the K_d value directly from the retardation factor (R_f) and soil properties (EPA, 1999).

$$R_f = \frac{m}{n_e} + \frac{\rho_b K_d}{n_e} \quad (6)$$

$$R_f = 1 + \frac{\rho_b K_d}{n_e} \quad (7)$$

$$R_f = 1 + \frac{\rho_b K_d}{n} \quad (8)$$

$$R_f = 1 + \frac{\rho_b K_d}{e} \quad (9)$$

where n , n_e , θ , and ρ_b are total porosity (cm^3 pore/ cm^3 total volume), effective porosity (cm^3 pore/ cm^3 total volume), volumetric water content in vadose zone (cm^3 water/ cm^3 total volume), and bulk density ($\text{g soil}/\text{cm}^3$ total volume), respectively.

R_f can be directly inserted into the transport code is one advantage of this method. This method also better approximates the physical conditions and chemical processes that occur in the field than a batch sorption experiment. A column experiment can be used to investigate both sorption and desorption reactions. Ideally, flow-through column experiments would be used exclusively for determining K_d values, but equipment cost, time constraints, experimental complexity, and data reduction uncertainties discourage more extensive use (EPA, 2004).

2.3. In-Situ Batch Method

The procedure for the in-situ batch method is similar to that for the laboratory

batch K_d method. This method requires the collection of paired soil and groundwater samples directly from the modeled aquifer system and the analyze of the amount (concentration) of heavy metal in the solid and liquid phases. The aqueous and solid phases are separated by centrifugation or filtration, and the solute concentration, C_p is then determined. The solid is analyzed to determine the concentration of A_p , the contaminant associated with the solid phase.

The advantage of this method over the laboratory K_d method is that it uses precise solution chemistry and solid phase mineralogy for modeling. However, this method is rarely used due to the analytical difficulties associated with measuring the exchangeable fraction of heavy metal in the solid phase.

2.4. Field Modeling Method

The field modeling method employs a transport model and existing groundwater monitoring data to estimate the K_d values of heavy metals. The minimum information required for such a calculation is the heavy metal concentrations at the source term, the date of release, the groundwater flow path, the groundwater flow rate, the heavy metal concentrations at a monitoring well, the distance between the source-release and the monitoring well, the dispersion coefficient, and the source term. The chemical retardation is then calculated as the ratio of pore-water velocity to heavy metal velocity (Equation (5)). Darcy's law can be used to calculate the pore-water velocity, v_p (EPA, 1999):

$$V_p = \frac{V_d}{n_e} \quad (10)$$

where V_d and n_e are Darcy velocity and effective porosity, respectively.

Field studies can provide precise estimates of contaminant time of travel because dissolved heavy metal concentrations are measured directly from monitoring well samples. The study site's exact geochemical and flow conditions are used to calculate K_d . The significant disadvantage of this technique is that it requires numerous water flow assumptions, such as uniform flow, direction, and path length, all of which affect the calculated K_d value. The calculated K_d values should not be used for contaminant transport calculations at other sites because they are model-dependent and highly site-specific.

2.5. K_{oc} Method

The K_{oc} method is less commonly used (EPA, 2004). Sorption of an organic/inorganic contaminant, such as polynuclear aromatic hydrocarbon (PAH) and heavy metal, is assumed to occur only on organic material in the soil for this method. The partitioning of the solid and solution phases is expressed as follows:

$$K_d = K_{oc} f_{oc} \quad (11)$$

where K_{oc} and f_{oc} are the ratios of the contaminant concentration on the organic matter on a dry weight basis to its dissolved concentration in the surrounding fluid (ml/g) and a fraction of organic carbon in the soil (mg/mg), respectively.

Moreover, the commonest correlation for K_{oc} is with the octanol-water partition coefficient (K_{ow}). A simplified relationship between these two parameters is given by Equation (12).

$$K_{oc} = \alpha K_{ow} \quad (12)$$

where α is a correlation coefficient (unitless).

The advantages of the K_{oc} method include a reasonably accurate indirect method, the ability to obtain K_{oc} values using look-up tables, f_{oc} is a simple measurement, and K_{oc} can be correlated with K_{ow} , which has been measured for many different chemicals. The main disadvantage of the K_{oc} method is that it can only estimate organic compound partitioning.

3. Method of Data Collection for Analysis

The Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) was used to increase the transparency of the collected data. “PRISMA is a protocol to conduct systematic reviews consisting of a 27-item checklist and a four-phase flow diagram (Figure 1), which was developed with the intent to increase the transparency and accuracy of literature reviews” (Kim et al., 2018; Pahlevan-Sharif et al., 2019). It has recently been applied to the environmental sciences via systematic meta-analysis (Zakari et al., 2021).

In this study, the PRISMA method was applied to summarize the K_d values of five heavy metals used in the existent literature as a reliable indicator for assessing the remediation process of toxic metals in soil, sediment, and groundwaters. The following keywords were used to search two databases (Web of Science and Google Scholar databases) for articles: “Distribution coefficients (K_d)*(Topic) AND Heavy Metal*(Topic) AND soil* (Topic) AND sediment*(Topic)”, “Distribution coefficients (K_d)*(Topic) AND Heavy Metal*(Topic)”, “Distribution coefficients (K_d)*(Topic) AND Cadmium*(Topic)”, “Distribution coefficients (K_d)*(Topic) AND Lead*(Topic)”, “Distribution coefficients (K_d)*(Topic) AND Copper*(Topic)”, “Distribution coefficients (K_d)*(Topic) AND Zinc*(Topic)”, and “Distribution coefficients (K_d)*(Topic) AND Arsenic*(Topic)”.

In Google Scholar, an advanced search combined the keywords mentioned above were used. For the first step of the screen, we used the following code: “Include = 1”, “Exclude = 0”, “Duplicate = 2”, and “Same K_d data = 3”, plus the comment to explain the reason. Then we look over the abstracts, which are usually available online. Only English research papers conducted to investigate K_d values of heavy metals sorption/desorption in soil/sediment were chosen. Abstracts of articles that showed promise for providing K_d values were kept and carefully screened in full texts before K_d values were recorded. The K_d experimental conditions, sorbing materials, aquatic medium, and metal initial concentration of the studied metal were then entered into a Microsoft Excel spreadsheet for compilation and analysis. Figure 2 depicts a summary of the article selection process. The database searches yielded 808 records (736 articles from the Web of Science databases and 72 articles from Google Scholar), where 594 were eliminated

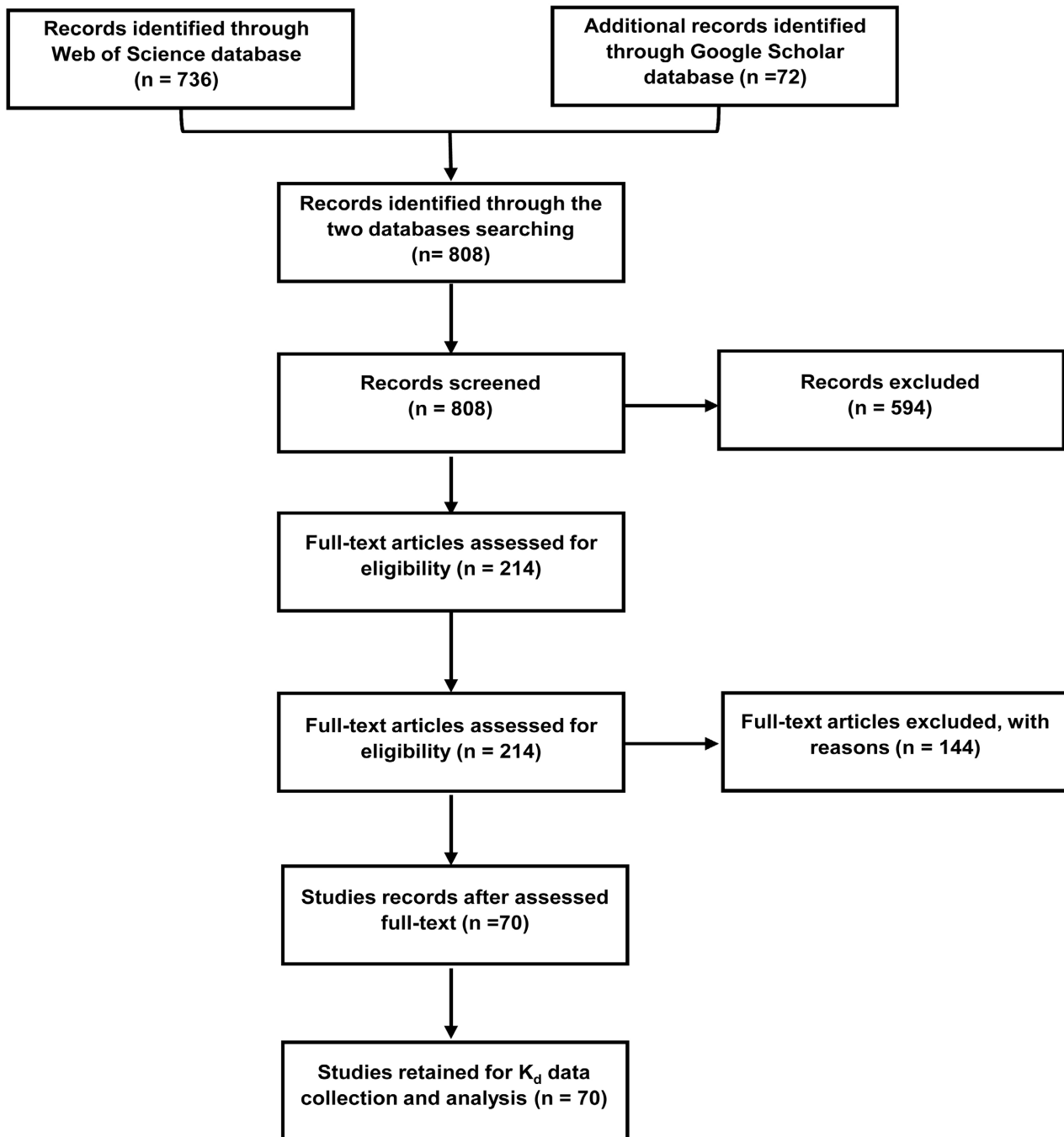


Figure 1. PRISMA Flow chart of the study selection process.

during the first screening stage. The full texts of the remaining 214 reviews were carefully read, and 144 were rejected because they did not meet the eligibility criteria. Finally, 70 papers were chosen.

4. Results

4.1. Reported K_d Values

Here, we provide a subsequent compilation of K_d values (Tables 1-5) on five

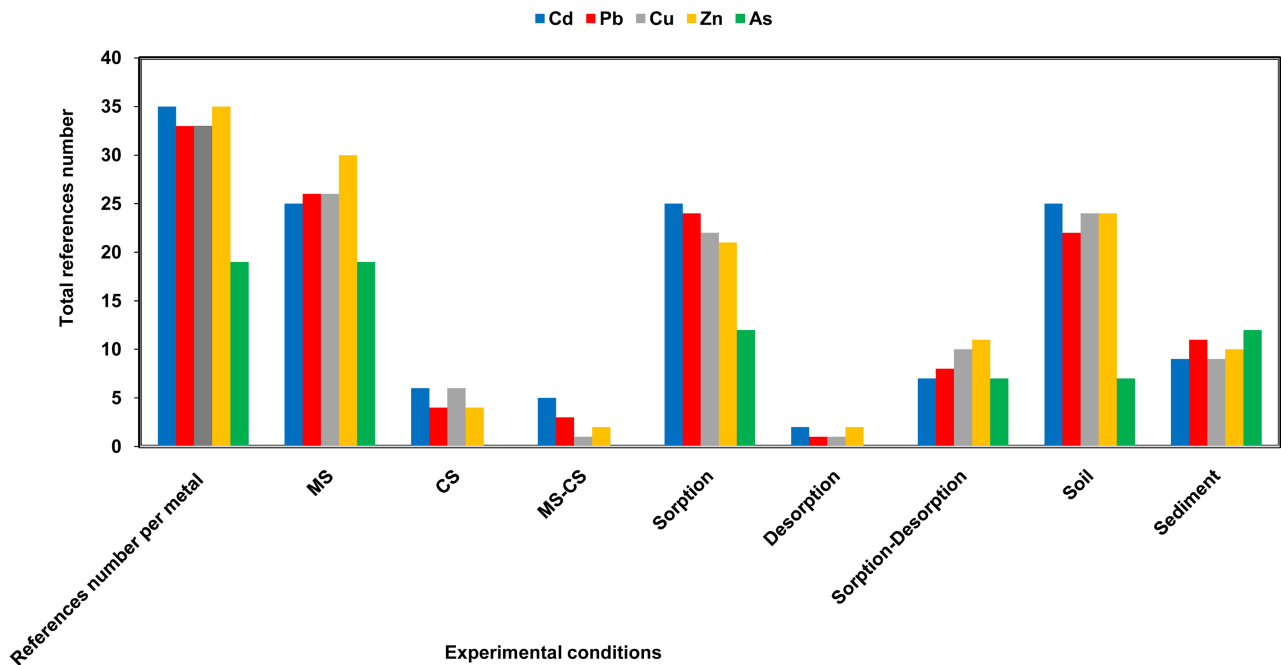


Figure 2. Characteristics of the number of K_d data observations in different references, sorption systems (MS, CS), conditions of liquid-solid exchange (sorption, desorption), and environmental components (soil, sediment) for the five heavy metals.

heavy metals (As, Pb, Cd, Cu, and Zn). Empty places in the table represent the missing data. In the tables, italic and bold values, bold and nonitalic values, and nonbold and nonitalic values represent the log values, the mean values, and the nonconverted values of K_d collected. A considerable amount of literature has been published on the determination, role, importance, and influencing factors of heavy metal K_d values in soil and sediments worldwide (Alloway, 2013; Nabelkova, 2012).

4.1.1. Cadmium (Cd)

Cd, in the +2-oxidation state, is considered as a potential environmental contaminant. Previous research has shown that organic matter, iron oxides, pH, and experimental reaction time all influence Cd sorption K_d values (Bielska et al., 2017; Ea & Grunzke, 2016). Diagboya et al. (2015) investigated the effects of organic matter and iron oxides on Cd retention and redistribution over time in batch competitive sorption experiments (from 1 to 90 days). On the one hand, their findings showed that removing organic matter resulted in a 33% decrease in K_d values on the first day. In contrast, K_d increased by nearly 100% in 7 days and more than 1000% in 90 days. They reported that the enhanced K_d values indicated that sorption occurred on the long run-on surfaces, which were masked by organic matter. On the other hand, removing iron oxides caused selective increases in the K_d values, dependently on the dominant soil constituent (s) in the absence of iron oxides. The K_d values of the iron oxides degraded samples nearly remained constant irrespective of aging, indicating that sorption on soil components other than the iron oxides is nearly instantaneous while iron oxides played

Table 1. Cadmium K_d data set for soil/sediment under various experimental conditions.

Experimental conditions	Sorbing materials	Aquatic medium	K_d Values (L/Kg)	Metal initial concentration	References
Batch method (solid-solution ratio: 1: 10; pH: (4.80 - 7.50); contact time: 6 days at 25°C)	Soil (Mono-metal Experiment)	Water	0.61	0 - 100 mg/L	(Oladipupo Azeez et al., 2018)
	Soil (Multi-metal Experiment)		0.44		
Batch method (solid-solution ratio: 1: 20; pH: 5; contact time: 6 h)	Soil	Water	15.9 - 932	10 - 200 mg/L	(Baghenejad et al., 2016)
K_d values were calculated directly from the retardation factor (R) and soil properties.	loamy sand media		300	Mean C_0 ($\mu\text{g/L}$)-(0.4)	(Behbahani et al., 2020)
Batch method (solid-solution ratio: 5: 45; contact time: 72 h)	Artificial soil (AS)		17.9 - 189.8	0.2 mg/ml	(Bielska et al., 2017)
Batch method (solid-solution ratio: 1:10; pH: 5.5 - 6.0; contact time: 24 h)	Soil	Water	6 - 2471.9	1000-mg-L ⁻¹	(Braz et al., 2013)
Batch competitive sorption method (solid-solution ratio: 1:20; pH: 3-7; contact time: 30 - 1440 min; 26°C)	Untreated soil	Water	0 - 0.02	(200 mg/L)	(Diagboya et al., 2015)
	IO-removed soil		0 - 1.85		
	OM-removed soil		0.01 - 1.2		
Batch method (solid-solution ratio: 1:10; pH: 7.4; contact time: 72 h)	Aquifer sediments	Water	44 - 1197		(Jakomin et al., 2015)
Batch desorption method (solid-solution ratio: 5:25; pH: 7.4; contact time: 48 h; 25°C)	Surface soil	Water	60 - 395		(Jalali & Hoursersesht, 2017)
	Surface soil		19 - 13921		
	Surface soil		97 - 425		
	Surface soil		21 - 430		
Batch desorption method (solid-solution ratio: 2:50; pH: 5; contact time: 22 h; 25°C)	Soil (single metal systems)	Water	90 - 184.5		(Jiang et al., 2012)
	Soil (binary metal systems)		85.84 - 110.5		
Batch adsorption method (solid-solution ratio: 2:24; pH: 1.2 - 8.8; contact time: 4 h; 28°C)	Soil	Water	21.1 - 1049.8	0.04 - 1.13 mmol/l	(Kim, 2014)

Continued

Batch adsorption method (solid-solution ratio: 2:50; pH: 5.5; contact time: 20 h; 25 °C)	Soil (single metal systems)	Water	117.4 - 348.6	50 - 1000 µmol/L	(Li et al., 2012)
	Soil (binary metal systems)		109.8 - 206.5		
Batch experiment (pH: 7.99 - 8.35)	Surface water sediments	Water	28.92		(Liao et al., 2021)
	Pore water sediments		21.56		
Batch experiment (pH: 7.22 - 8.56)	sediment	Water	24,000 - 140,000		(Liu et al., 2019)
diffusive gradient in thin-film (DGT) technique	Plated soils (10 g/kg sewage sludge)	Water	168.7 - 291.9		(Mohseni et al., 2020)
Plated soils (30 g/kg sewage sludge)			416.1 - 1271.1		
Batch equilibrium adsorption (solid-solution ratio: 2.5:25; acidic pH (4.44), alkaline pH (7.77); contact time: 4 h)	Soils	Water	11.13 - 1673.72	10 - 800 µg/l	(Rezaei et al., 2021)
	Soils		7.82 - 376.05	1 - 1000 mg/l	
Batch adsorption experiment (solid-solution ratio: 3:30; pH (5.1 to 7.9); contact time: 24 h; 20 °C)	Soils		5.8 - 4885.5	28.1 - 2224.8 mg/L	(Shaheen et al., 2015)
	Soils		4.3 - 7334.8		
Review report	Soils (mono metal sorption system)		7.3 - 1699.6		(Shaheen et al., 2013)
	Soils in competitive sorption system)		3.1 - 1588.3		
Batch desorption method (solid-solution ratio: 3:30; alkaline (pH 7.87), acidic (pH 4.95); contact time: 24 h; 20 °C)	Amended Alfisol1		86.5 - 79.9		(Shaheen et al., 2018)
	Amended Alfisol2	Water	372.3 - 123	50 - 300 mg/L	
	Amended Alfisol3		1476.4 - 323.7		
	Amended Entisol1		4.8 - 17.3		
	Amended Entisol2		41.9 - 44.3		
	Amended Entisol3		32.7 - 158.8		
Batch equilibrium adsorption experiment (solid-solution ratio: 2:20; pH: 6.5; contact time: 24 h)	Topsoil		Deionized water		7 - 14,339
Batch equilibrium adsorption experiment (solid-solution ratio: 1:20; pH: 2 - 9; contact time: 24 h)	control soil	Water	1.7 - 1412.5		(Tahervand & Jalali, 2017)

Continued

	bentonite-amended soil		2.7 - 1862.1		
	calcite-amended soil		2.6 - 1479.1		
	zeolite-amended soil		2.8 - 1443.5		
Batch equilibrium adsorption experiment (solid-solution ratio: 2:50; contact time: 24 h)	Acidic soil (pH: 4 - 7)	Water	23 - 675	0.05 - 10 meq/L	(Venegas et al., 2015)
	Treated soil		3.0 - 2180		
Batch method (solid-solution ratio: 25:50; contact time: 2 h; 25 °C)	Soil	Water	3.24 - 5145		(Yang et al., 2017)
Batch method	profile soil		102 - 103		(Yuan et al., 2017)
Batch method	lacustrine sediment		162.804 - 294.153		(Yuan et al., 2020)
Batch method	Sediment (Jan. 2014)	Deionized water	4.95 - 5.37		(Zhang et al., 2018)
	Sediment (Jan. 2014)		5.29 - 5.79		
	Sediment (whole 2014)		4.95 - 5.79		
Batch-type desorption experiment (solid-solution ratio: 1:10; contact time: 2 h)	Calcareous soils	Water	387.8 - 283.8	50 - 400 mg/L	(Jalali & Vafaei, 2017)
	Calcareous soils		256.3 - 183.6	51 - 400 mg/L	
Batch method	Lake sediments	Deionized water	2.3 - 2.9		(Li et al., 2017)
Batch experiment	Sediment		$7.69 \times 10^1 - 1.50 \times 10^4$		(Boyer et al., 2018)
	Stream bed sediment		<3		(Sedeño-Díaz et al., 2019)
Batch equilibration method (solid-solution ratio: 1:25; contact time: 24 h; 25 °C)	River Sediments	Deionized water	90.26 - 523.41	100 - 10 mg/L	(Fan et al., 2017)
Batch equilibration method (solid-solution ratio: 5:50; contact time: 1 h - 48 h)	Alluvial sediment		0.7674	60 mg·L ⁻¹	(de Oliveira Barros et al., 2021)
Batch experiment (solid-solution ratio: 2.5:50; contact time: 24 h; 25 °C)	Sand-Attapulgate Cutoff Wall Backfill Media		$10^1 - 10^4$		(Zhang et al., 2021)
Batch equilibration technique (solid-solution ratio: 3:30; contact time: 1 and 7 days)	Soil		5.21 - 380	25 - 150 mg/L	(Elbana et al., 2018)
Batch adsorption experiments (solid-solution ratio: 1:20; pH: 5.28; contact time: 24 h, 25 °C)	Untreated alluvial soil		21.8 - 62.6	10 - 200 mg/L	(Ren et al., 2020)

Continued

	Removal OM	20.6 - 47.6	
	Removal Fed	290.2 - 537.7	
	Removal Mn	11.2 - 23.2	
Batch adsorption experiments (solid-solution ratio: 1:10; contact time: 24 h, 25 °C)	Untreated fluvial soil	7 - 217	(Shaheen et al., 2017)
	Biosolids-amended fluvial soil	7.6 - 193.7	50 - 300 mg/L
	Untreated Calcareous soil	3.6 - 149.1	
	Biosolids-amended calcareous soil	3.9 - 147.6	
Total		66	35

a greater role with time. The authors conclude that in the studied soils, organic matter content determines the immediate relative metal retention while iron oxides determine the redistribution of metals with time. Ren et al. (2020) investigated the effects of organic matter, free Fe oxides and Mn oxides on Cd adsorption in alluvial soil. They found a similar effect of organic matter and iron oxides on Cd retention and redistribution. The results showed that when organic matter and Mn oxides were removed from soils, the K_d values of Cd decreased by a maximum of 25.2% and 64.1%, respectively, when compared to untreated soils. Furthermore, unlike organic matter and Mn oxides, K_d values of Cd sorption increased by 1670.2% after free Fe oxides were removed. According to these findings, soil with a high organic matter content has strong Cd sorption, and the stability of Cd and organic complexes increases with pH. These interpretations are consistent with those reported by (Elbana et al., 2018), who used kinetic sorption batch methods to quantify Cd retention by ten soils over a wide range of Cd input concentrations. The estimated K_d values ranged from 5.21 to 380 L/kg. They found that soils with high organic matter and pH showed strong Cd sorption.

Several studies have attempted to explain the effect of the initial concentration of included Cd on K_d values. Most K_d values of Cd decreased as the initial concentration of the included Cd cation in the experiment solution increased. Baghenejad et al. (2016) investigated the competitive adsorption behavior of several important pollutant metals, including Cd. They found that K_d values of Cd decreased significantly from 3195 to 7.1, 2657 to 6.7, 211 to 2, 2865 to 4.6, 2244 to 1.8, and 64.8 to 0.8 L/kg as various added metal concentrations increased from 10 to 200 mg/L for different soils label 1 to 6. In another study, Rezaei et al. (2021) found that mean K_d values of Cd decreased from 376.05 to 7.82 L/kg only when the initial concentration of Cd was high and increased from 1 to 1000 mg/L. When the initial Cd added concentration was low, they found that the K_d

Table 2. Led K_d Data set for soil/sediment under various experimental conditions.

Experimental conditions	Sorbing materials	Aquatic medium	K_d Values (L/Kg)	Metal initial concentration	References
Batch method (solid-solution ratio: 2: 20; pH: 12; contact time: 24 h)	Loamy sand soil	Water	41.95 - 57.33	25 - 300 mg/L	(Al-Oud & Ghoneim, 2018)
	Sandy loam soil		52.04 - 67.15		
Batch method (solid-solution ratio: 1: 10; pH: (4.80 - 7.50); contact time: 6 days at 25°C)	Soil (Mono-metal Experiment)	Water	0.83	0 - 100 mg/L	(Oladipupo Azeez et al., 2018)
	Soil (Multi-metal Experiment)		0.58		
Batch method (solid-solution ratio: 1: 20; pH: 5; contact time: 6 h)	Soil	Water	1216 - 4975	10 - 200 mg/L	(Baghenejad et al., 2016)
K_d values were calculated directly from the retardation factor (R) and soil properties.	loamy sand media		2000	Mean C_0 ($\mu\text{g/L}$)-(0.4)	(Behbahani et al., 2020)
Batch method (solid-solution ratio: 1:10; pH: 5.5-6.0; contact time: 24 h)	Soil	Water	196.4 - 5572.5	1000 mg·L ⁻¹	(Braz et al., 2013)
Batch competitive sorption method (solid-solution ratio: 1:20; pH: 3-7; contact time: 30 - 1440 min; 26°C)	Untreated soil	Water	0.03 - 0.53	(200 mg/L)	(Diagboya et al., 2015)
	IO-removed soil		0.05 - 1.7		
	OM-removed soil		0.02 - 75.92		
Batch method (Desorption)	sediments	Water	4.01 - 5.74		(Huang et al., 2012)
Batch method (solid-solution ratio: 1:10; pH: 7.4; contact time: 72 h)	Aquifer sediments	Water	554 - 10,000		(Jakomin et al., 2015)
Batch desorption method (solid-solution ratio: 2:50; pH:5; contact time: 22 h; 25 C)	Soil (single metal systems)	Water	829.3 - 1107.5		(Jiang et al., 2012)
	Soil (binary metal systems)		514.8 - 732.3		
Batch desorption method (solid-solution ratio: 2:50; pH: 5.5; contact time: 20 h; 25 C)	Soil (single metal systems)	Water	508.5 - 1639.7	50 - 1000 $\mu\text{mol/L}$	(Li et al., 2012)
	Soil (binary metal systems)		315 - 1065.6		

Continued

Batch experiment (pH: 7.99 - 8.35)	Surface water sediments	Water	233.44		(Liao et al., 2021)
	Pore water sediments		168.85		
Batch experiment (pH: 7.22 - 8.56)	sediment	Water	240,000 - 7,000,000		(Liu et al., 2019)
Batch adsorption test (solid-solution ratio: 0.2:20; pH:8; contact time: 2 h; 25 C)	calcareous soil (single metal systems)		1841.1		(Mahzari et al., 2013)
	calcareous soil (binary metal systems: Pb-Cu)	Water	1652.9		
	calcareous soil (binary metal systems: Pb-Mn)		1702.4		
diffusive gradient in thin-film (DGT) technique	Planted soils (10 g/kg sewage sludge)	Water	738.8 - 414.5		(Mohseni et al., 2020)
	Planted soils (30 g/kg sewage sludge)		539.5 - 1413.8		
Batch desorption method (solid-solution ratio: 3:30; alkaline (pH 7.87), acidic (pH 4.95); contact time: 24 h; 20 C)	Amended Alfisol1		164.7 - 79.1		(Shaheen et al., 2018)
	Amended Alfisol2	Water	718.1 - 218.6	50 - 300 mg/L	
	Amended Alfisol3		2663.9 - 3913.5		
	Amended Entisol1		47.7 - 12.9		
	Amended Entisol2		80.5 - 34.4		
	Amended Entisol3		493.3 - 21.8		
Batch equilibrium adsorption experiment (solid-solution ratio: 2:20; pH: 6.5; contact time: 24 h)	Topsoil	De-ionized water	121 - 7020		(Soares et al., 2021)
Batch equilibrium adsorption experiment (solid-solution ratio: 2:50; contact time: 24 h)	Acidic soil (pH: 4 - 7)	Water	2140 - 63,100	0.05 - 10 meq/L	(Venegas et al., 2015)
	Treated soil		35 - 91,000		
Batch method	profile soil		10 ⁴ - 10 ⁶		(Yuan et al., 2017)
Batch method	lacustrine sediment		49.939 - 179.044		(Yuan et al., 2020)
Batch method	Sediment (Jan. 2014)	Deionized water	5.32 - 5.95		(Zhang et al., 2018)
	Sediment (Jan. 2014)		5.02 - 5.59		

Continued

	Sediment (whole 2014)		5.02 - 5.95		
Batch test	Sediment	Distilled water	4.23		(Yavar Ashayeri & Keshavarzi, 2019)
Batch method	Lake sediments	deionized water	4.1 - 4.7		(Li et al., 2017)
Batch adsorption experiment (solid-solution ratio: 2:100; pH: 5.5; contact time: 22h)	Soil		713 - 11	20 - 250 mg/L	(Ugochukwu et al., 2013)
Batch experiment (solid-solution ratio: 0.5:5; pH: 1.5; contact time: 24 h; 25°C)	Natural sediment		3.3 - 5.0	0.025 - 1 mol/L	(Wang et al., 2016)
			3.3 - 5.2		
Batch experiment	Sediment		$3.33 \times 10^1 -$ 5.60×10^6		(Boyer et al., 2018)
Batch experiment (solid-solution ratio: 2:50; contact time: 48 h)	Soil		10 - 339 624		(Janik et al., 2015)
Batch experiment (solid-solution ratio: 1:30; contact time: 0 - 170 h)	Coarse sand Soil	Ground water	7570 - 31,000		(Maity & Pandit, 2014)
	Fine sand Soil		6700 - 18,300		
Batch experiment (solid-solution ratio: 1:2; contact time: 72 h)	Soils		279 - 969	4 - 358 mg/kg	(Mrdakovic Popic et al., 2014)
Batch method	Soil		<0.002 - 33.30		(Egbi et al., 2015)
	Stream bed sediment		$3 < K_d$		(Sedeño-Díaz et al., 2019)
Batch equilibration method (solid-solution ratio: 1:20; contact time: 24 h)	calcareous soil		136.37	0 - 100 mg/L	(Al-Hayani & Al-Obaidi, 2019)
Batch equilibration method (solid-solution ratio: 2:40; contact time: 24 h; 298 K)	calcareous soil		111.8 - 255.5	100 - 0 mg/L	(Al-Obaidi & Al-Obadi, 2019)
Batch adsorption experiments (solid-solution ratio: 1:10; contact time: 24 h, 25°C)	Untreated fluvial soil		4563.4 - 6871.8	50 - 300 mg/L	(Sharma et al., 2017)
	Biosolids-amended fluvial soil		3994.6 - 7763.6		

Continued

	Untreated Calcareous soil	1128.2 - 5122.2	
	Biosolids-amended calcareous soil	2050.6 - 4318	
Total		56	33

values gradually increased with an increase of Cd added concentration. In this case, they found that the medium K_d values of Cd increase from 11.13 to 1673.72 L/kg when the initial low concentration of Cd increases from 10 to 800 $\mu\text{g/L}$. According to the authors, the Cd sorption was specific and high at low initial concentrations, and the K_d increased as these initial concentrations increased. At high concentrations, the specific sorption sites were gradually occupied with the increase of initial Cd concentrations, resulting in lower K_d values. It has been found that the higher K_d values of Cd obtained in experiments with lower metal concentrations are associated with the sorption sites of high selectivity (Loganathan et al., 2012). However, increasing rates of metals' addition to soils may result in saturation of sorption sites for Cd in soils, thereby decreasing the sorption capacity.

The sorption systems are also well-known for significantly influencing Cd's K_d values. Li et al. (2012) performed batch equilibrium experiments in paddy soils using MS and CS solutions and discovered that K_d values of Cd in MS were higher than K_d values of Cd in CS. Similar results were reported by Shaheen et al. (2015), who conducted batch experiments to investigate Cd sorption characteristics in MS and CS. The mean's K_d values of Cd were 1588.3 L/kg and 1699.6 L/kg for MS and CS. They found that K_d values of Cd decrease in CS compared to MS. Under the CS, the mean K_d value of Cd (0.44 L/kg) was found to be lower than the other metals in the studied soils, indicating that Cd was less retained in soil than other metals (Baghenejad et al., 2016). Under CS conditions, it is implied that Cd may pose a greater threat to plants and groundwater than other metals because competition for the same available sorption sites tends to suppress the strength and magnitude of heavy metal retention when more than one heavy metal is present in a soil system.

4.1.2. Lead (Pb)

Pb occurs naturally in all environmental media, including air, soil, sediment, and water. It is not a required element for life. Pb contamination of air, soil, sediment, and water is considered a risk to human health, plant growth, and development. The fate and transportation of Pb ions in the environment are generally controlled by sorption and desorption. Several studies have used the K_d to evaluate the sorption and desorption of Pb on soil solids and liquid interfaces (Huang et al., 2012; Maity & Pandit, 2014). Although sewage sludge is beneficial as organic fertilizer, it has also been shown that it affects metal desorption by increasing metal loading and inducing chemical changes in soil and sediment over

Table 3. Copper K_d data set for soil/sediment under various experimental conditions.

Experimental conditions	Sorbing materials	Aquatic medium	K_d Values (L/kg)	Metal initial concentration	References
Batch method (solid-solution ratio: 0.5:50; pH: (2 - 8); contact time: (5 - 300 min))	clay	Ultrapure water	$0 < K_d < 150$	40 - 2000 mg/L	(Alandis et al., 2019)
Batch method (solid-solution ratio: 1:10; pH: (4.80 - 7.50); contact time: 6 days; 25 °C)	Soil (Mono-metal Experiment)	Water	0.43	0 - 100 mg/L	(Oladipupo Azeez et al., 2018)
	Soil (Multi-metal Experiment)		0.57		
Batch method (solid-solution ratio: 1:20; pH: 5; contact time: 6 h)	Soil	Water	1235 - 3227	10 - 200 mg/L	(Baghenejad et al., 2016)
Batch method (solid-solution ratio: 0.4:40; pH: 5; contact time: 6 h)	Shekarbani soil series	Water	9.0 - 3010	20 - 2000 mg/L	(Baghernejad et al., 2014)
	Sepidan soil series		42 - 7495		
K_d values were calculated directly from the retardation factor (R) and soil properties.	loamy sand media		1000	Mean C_0 ($\mu\text{g/L}$)-(68)	(Behbahani et al., 2020)
Batch method (solid-solution ratio: 1:20; pH: 7.5 - 8.55; contact time: 8 h)	River sediments (BRS1)	Milli-Q water	1.08		(Borah et al., 2018)
Batch method (solid-solution ratio: 1:10; pH: 5.5 - 6.0; contact time: 24 h)	Soil	Water	50.5 - 7368.7	1000 mg·L ⁻¹	(Braz et al., 2013)
Batch competitive sorption method (solid-solution ratio: 1:20; pH: 3 - 7; contact time: 30 - 1440 min; 26 °C)	Untreated soil	Water	0.01 - 0.17	(200 mg/L)	(Diagboya et al., 2015)
	IO-removed soil		0.00 - 8.33		
	OM-removed soil		0.14 - 1.70		
Batch method (Desorption)	sediments	Water	3.58 - 5.41		(Huang et al., 2012)
Batch desorption method (solid-solution ratio: 5:25; pH: 7.4; contact time: 48 h; 25 °C)	Calcareous Soils	Water	72 - 3100		(Jalali & Vafaei, 2017)
	Calcareous Soils		169 - 13,021		
	Calcareous Soils		332 - 10,727		
	Calcareous Soils		189 - 6888		
Batch desorption method (solid-solution ratio: 5:500; pH: 4; contact time: 24 h)	Soil (SA)		0.6 - 5.31	30 - 100 mg/L	(Jalayeri et al., 2015)
	Soil (SE)		0.55 - 3.41		
Batch experiment (pH: 7.99 - 8.35)	Surface water sediments	Water	68.46		(Liao et al., 2021)
	Pore water sediments		55.06		

Continued

Batch experiment (pH: 7.22 - 8.56)	sediment	Water	32,000 - 80,000		(Liu et al., 2019)
Batch adsorption test (solid-solution ratio: 0.2:20; pH: 8; contact time: 2 h; 25 °C)	calcareous soil (single metal systems)	Water	5517.3		(Mahzari et al., 2013)
Batch desorption method (solid-solution ratio: 3:30; alkaline (pH 7.87), acidic (pH 4.95); contact time: 24 h; 20 °C)	amended Alfisol1		82 - 46.9		(Shaheen et al., 2018)
	amended Alfisol2	Water	231.9 - 117.3	50 - 300 mg/L	
	amended Alfisol3		1116.6 - 453.1		
	amended Entisol1		64.3 - 26.1		
	amended Entisol2		70.2 - 2.3		
	amended Entisol3		81.3 - 9.7		
Batch equilibrium adsorption experiment (solid-solution ratio: 2:20; pH: 6.5; contact time: 24 h)	Topsoil (A horizon, 0 - 20 cm)	De-ionized water	105 - 4598		
Batch equilibrium adsorption experiment (solid-solution ratio: 1:20; pH: 2 - 9; contact time: 24 h)	control soil		1.1 - 4633.5		(Tahervand & Jalali, 2017)
	bentonite-amended soil	Water	5.9 - 6929.5		
	calcite-amended soil		4.5 - 6592.6		
	zeolite-amended soil		5.6 - 4750.0		
Batch equilibrium adsorption experiment (solid-solution ratio: 2:50; contact time: 24 h)	Acidic soil (pH: 4 - 7)	Water	370 - 2400		(Venegas et al., 2015)
	Treated soil		9.0 - 4860		
Batch method	profile soil		$10^3 - 10^4$		(Yuan et al., 2017)
Batch method	lacustrine sediment		268.917 - 826.688		(Yuan et al., 2020)
Batch method	Sediment (Jan. 2014)	Deionized water	3.49 - 3.95		(Zhang et al., 2018)
	Sediment (Jan. 2014)		4.03 - 4.51		
	Sediment (whole 2014)		3.49 - 4.51		
Batch-type desorption experiment (solid-solution ratio: 1:10; contact time: 2 h)	Calcareous soils	Water	1766.2 - 4317.9	50 - 400 mg/L	(Jalali & Hoursersht, 2017)
	Calcareous soils		2029.6 - 3704.8	51 - 400 mg/L	
Batch experiment	Sediment		$4.40 \times 10^0 - 2.94 \times 10^5$		(Boyer et al., 2018)

Continued

Batch experiment (solid-solution ratio: 2:50; contact time: 48 h)	Soil		23 - 8589		(Janik et al., 2015)
Laboratory batch experiments (solid-solution ratio: 5:50; contact time: 24 h)	Oxisol	Water	3.18 - 3.35		(Aishah et al., 2018)
	Ultisol		3.23 - 3.42		
Batch experiment	soil		115 - 1145		(Christiansen et al., 2015)
Batch method	Soil		43 - 1.8		(Egbi et al., 2015)
	Stream bed sediment		$3 < K_d < 4$		(Sedeño-Díaz et al., 2019)
Batch equilibration method (solid-solution ratio: 1:25; contact time: 24 h; 25 °C)	River Sediments	Deionized water	162.5 - 1579.8	100 - 10 mg/L	(Fan et al., 2017)
Batch equilibration method (solid-solution ratio: 5:50; contact time: 1 h - 48 h)	Alluvial sediment		0.8593	100 mg/L	(Zhang et al., 2021)
Batch equilibration method (solid-solution ratio: 2:50; contact time: 24 h; 298 K)	Soil		8316 - 14,476	60 - 180 mg/L	(Al-Hassoon et al., 2019)
Batch sorption experiments (solid-solution ratio: 0.5:25; contact time: 24 h at 25 °C)	Soil		20 - 76,460	160 mg/L	(Ding et al., 2018)
Batch adsorption experiments (solid-solution ratio: 1:10; contact time: 24 h, 25 °C)	Untreated fluvial soil		825.1 - 24935.2		(Shaheen et al., 2017)
	Biosolids-amended fluvial soil		930.2 - 17784.5	50 - 300 mg/L	
	Untreated Calcareous soil		36.6 - 10205.4		
	Biosolids-amended calcareous soil		427.5 - 7047.7		
Total			58		33

time (Huang et al., 2020; Khan et al., 2015). In this regard, Mohseni et al. (2020) reported that K_d values of Pb desorption increased from 539.5 to 1413.8 L/kg over incubation time in planted soils treated with 30 g/kg of sewage sludge. The K_d values of Pb desorption decreased from 738.8 to 414.5 L/kg in soils treated with 10 g/kg sewage sludge during the incubation period, indicating that applying a high sewage sludge rate to soil may immobilize Pb and reduce its solubility. These findings support the findings of (Venegas et al., 2015) regarding the retention of Pb in Alfisol and Entisol after biosolid application. They discovered that

Table 4. Zinc K_d data set for soil/sediment under various experimental conditions.

Experimental conditions	Sorbing materials	Aquatic medium	K_d Values (L/Kg)	Metal initial concentration	References
Batch method (solid-solution ratio: 01:10; pH: (7.4 - 8.6); contact time: 24 h)	Untreated soils	Water	363.1 - 4885.5	0.5 - 6.0 mmol/L	(Antoniadis et al., 2018)
	CaCO ₃ -removed soils		13 - 143.2		
	Oxides-removed soils		3.1 - 126.5		
Batch method (solid-solution ratio: 1:10; pH: (4.80 - 7.50); contact time: 6 days at 25°C)	Soil (Mono-metal Experiment)	Water	0.8	0 - 100 mg/L	(Oladipupo Azeez et al., 2018)
	Soil (Multi-metal Experiment)		0.71		
Batch method (solid-solution ratio: 0.1:0.05; pH: (6 - 7.88); contact time: (1 - 48 h) at 20°C)	Untreated soil	Water	280 - 5560	10 - 120 mg/L	(Azouzi et al., 2015)
	OM-removed soil		4060 - 13,580		
	Untreated		12,450 - 166,160		
	Untreated soil		860 - 5870		
	OM-removed		190 - 610		
Batch method (solid-solution ratio: 1:20; pH: 5; contact time: 6 h)	Untreated soil	Water	4810 - 11,730	10 - 200 mg/L	(Baghernejad et al., 2016)
	Soil		84.9 - 1630		
Batch method (solid-solution ratio: 0.4:40; pH: 5; contact time: 6 h)	Shekarbani soil series	Water	6 - 1275	20 - 2000 mg/L	(Baghernejad et al., 2014)
	Sepidan soil series		39 - 5108		
K_d values were calculated directly from the retardation factor (R) and soil properties.	loamy sand media		400	Mean C ₀ (µg/L)-(68)	(Behbahani et al., 2020)
Batch method (solid-solution ratio: 1:20; pH: 7.5 - 8.55; contact time: 8 h)	River sediments (BRS1)	Milli-Q water	0.64		(Borah et al., 2018)
Batch method (solid-solution ratio: 1:10; pH: 5.5 - 6.0; contact time: 24 h)	Soil	Water	1.4 - 7933	1000-mg·L ⁻¹	(Braz et al., 2013)
Batch method (solid-solution ratio: 1:10; pH: 7.4; contact time: 72 h)	aquifer sediments	Water	114 - 5633		(Jakomin et al., 2015)
			131 - 10,160		
Batch method (solid-solution ratio: 2:24)	Soil	Water	19.9 - 7739.3	0.04 - 1.13 mmol/l	(Kim, 2014)
Batch experiment (pH: 7.99 - 8.35)	Surface water sediments	Water	61.7		(Liao et al., 2021)

Continued

	Pore water sediments		48.92		
Batch experiment (pH: 7.22 - 8.56)	sediment	Water	$9.4 \times 10^4 - 8.5 \times 10^5$		(Lin et al., 2017a)
diffusive gradient in thin-film (DGT) technique	Plated soils (10 g/kg sewage sludge)	Water	766.4 - 320.2		(Mohseni et al., 2020)
	Plated soils (10 g/kg sewage sludge)		979.7 - 1814.		
Batch experiment (solid-solution ratio: 3:30; pH (5.1 to 7.9); contact time: 24 h; 20°C)	Soils (mono metal sorption system)		7.3 - 1699.6	28.1 - 2224.8 mg/L	(Shaheen et al., 2015)
	Soils in competitive sorption system)		3.1 - 1588.3		
Review report	Soils (mono metal sorption system)		12.4 - 4885.5		(Shaheen et al., 2013)
	Soils in competitive sorption system)		9.6 - 7334.8		
Batch desorption method (solid-solution ratio: 3:30; alkaline (pH 7.87), acidic (pH 4.95); contact time: 24 h; 20°C)	Amended Alfisol1		59.4 - 125		(Shaheen et al., 2018)
	Amended Alfisol2	Water	137.5 - 187.5	50 - 300 mg/L	
	Amended Alfisol3		293.8 - 475		
	Amended Entisol1		2.3 - 8.7		
	Amended Entisol2		70.9 - 1.6		
	Amended Entisol3		17.4 - 21.7		
Batch equilibrium adsorption experiment (solid-solution ratio: 2:20; pH: 6.5; contact time: 24 h)	Topsoil (A horizon, 0 - 20 cm)		Deionized water		5 - 123849
Batch equilibrium adsorption experiment (solid-solution ratio: 1:20; pH: 2 - 9; contact time: 24 h)	control soil		1.7 - 3993.1		(Tahervand & Jalali, 2017)
	bentonite-amended soil	Water	7.6 - 4760.0		
	calcite-amended soil		12.5 - 10637.6		
	zeolite-amended soil		7.9 - 7655.1		
Batch equilibrium adsorption experiment (solid-solution ratio: 2:50; contact time: 24 h)	Acidic soil (pH: 4 - 7)		Water	32 - 2345	0.05 - 10 meq/L
	Treated soil	2.0 - 2660			

Continued

Batch method	profile soil		$10^3 - 10^4$	(Yuan et al., 2017)
Batch method	lacustrine sediment		202.843 - 1752.973	(Yuan et al., 2020)
Batch method	Sediment (Jan. 2014)	Deionized water	4.15 - 4.61	(Zhang et al., 2018)
	Sediment (Jan. 2014)		3.81 - 4.55	
	Sediment (whole 2014)		3.81 - 4.61	
Batch test	Sediment	Distilled water	4.22	(Yavar Ashayeri & Keshavarzi, 2019)
Batch-type adsorption experiment (solid-solution ratio: 2:20; pH: 5.5; contact time: 24 h; 25°C)	Soil	Water	1.4 - 7933.0	(Braz et al., 2013)
Batch-type desorption experiment (solid-solution ratio: 1:10; contact time: 2 h)	Calcareous soils	Water	1860.7 - 2051.1 50 - 400 mg/L	(Jalali & Hoursersesht, 2017)
			2455.3 - 1918.3 51 - 400 mg/L	
Batch method	Lake sediments	deionized water	3.4 - 4.6	(Li et al., 2017)
Batch experiment	Sediment		$2.11 \times 10^0 - 1.71 \times 10^4$	(Boyer et al., 2018)
Batch experiment (solid-solution ratio: 2:50; contact time: 48 h)	Soil		2 - 20 276	(Janik et al., 2015)
Laboratory batch experiments (solid-solution ratio: 5:50; contact time: 24 h)	Oxisol	Water	2.62 - 2.81	(Aishah et al., 2018)
	Ultisol		2.68 - 2.82	
Batch method	Soil		2.04 - 6.5	(Egbi et al., 2015)
Batch method	Soil		3.75 - 8.88	(Sharma & Sharma, 2013)
		Deionized water	4.35 - 12.02	
			5.99 - 22.15	
			3.25 - 6.68	
			2.08 - 5.62	
Batch method	Soils		4.35 - 12.02	(Gurpreet et al., 2012)
		Deionized water	3.38 - 7.25	
			4.55 - 6.67	
			2.88 - 5.64	

Continued

	Stream bed sediment	$3 < K_d < 4$		(Sedeño-Díaz et al., 2019)
Batch adsorption experiments (solid-solution ratio: 1:10; contact time: 24 h, 25°C)	Untreated fluvial soil	16 - 1257.9		(Shaheen et al., 2017)
	Biosolids-amended fluvial soil	16.1 - 1334.9	50 - 300 mg/L	
	Untreated Calcareous soil	6.2 - 523.9		
	Biosolids-amended calcareous soil	7 - 490.2		
Batch equilibration method	Soil	90.56 - 2.23	0 - 150 mg/ml	(Das & Das, 2015)
Total		74		35

after three biosolid application rates (20, 50, and 100 Mg/ha), the K_d values of Pb sorption increased from 79.1 to 164.7, 218.6 to 718.1, and 2663.9 to 3913.5 L/kg in Alfisol. After high biosolid application rates (50 and 100 Mg/ha), the K_d values of Pb sorption increased from 34.4 to 80.5 and from 21.8 to 493.3 L/kg in the case of Entisol. They hypothesized that this would immobilize Pb and reduce its solubility. In contrast, applying a low biosolids rate of 20 Mg/ha to Entisol decreased the K_d values of Pb sorption from 47.7 to 12.9 L/kg, which may mobilize Pb, increase its solubility, and improve phytoextraction.

There have been numerous attempts to explain the effect of soil type and texture, ionic strength, and reaction temperature on K_d values. In batch adsorption experiments, Ugochukwu et al. (2013) investigated the effect of soil type and inorganic ions on Pb adsorption on three acidic soils, including yellow-brown soil (YBS), latosol soil (LS), and lateritic red soil (LRS). The reported K_d value of Pb was found to be highest (713 L/kg) in YBS soil and the lowest (11 L/kg) in LRS soil. Regarding the effect of ionic strength on K_d values, they found that K_d values of Pb^{2+} decreased from 1688 and 190 L/kg to 747 and 87 L/g due to an increase in the ionic strength of K^+ and Ca^{2+} from 0.001 mol/L to 0.1 mol/L. Wang et al. (2016) also found similar results, and they reported that the K_d values of Pb decreased from 5.0 - 3.6 to 4.2 - 3.5 at $T = 25^\circ C$ and from 5.2 - 3.8 to 4.1 - 3.5 at $T = 15^\circ C$ when the ionic strengths increased from 0.005 to 0.05 Mol, respectively.

It has been reported that sediments are a sink for heavy metals because the sediments in water bodies are usually found with high heavy metal concentrations when compared with the surrounding surface soil (Huang et al., 2020; Zhuang & Gao, 2013). As a result, many studies have investigated the influencing factors of Pb K_d values in sediments to assess the sorption and desorption of Pb in deposited sediments. Therefore, in lacustrine sediments, Yuan et al. (2020) found that K_d values of Pb desorption ranged from 49.939 to 179.044 L/kg. They

Table 5. Arsenic K_d data set for soil/sediment under various experimental conditions.

Experimental conditions	Sorbing materials	Aquatic medium	K_d Values (L/kg)	Metal initial concentration	References
K_d values were calculated directly from the retardation factor (R) and soil properties.	loamy sand media		500	Mean C_0 ($\mu\text{g/L}$)-(68)	(Behbahani et al., 2020)
Batch method (solid-solution ratio: 1:20; pH: 7.5 - 8.55; contact time: 8 h)	River sediments (BRS1)	Milli-Q water	1.32		(Borah et al., 2018)
Batch method (solid-solution ratio: 2:20; contact time: 24 h)	Soil	Water	1.38 - 21.6		(Kandakji et al., 2015)
Batch method	profile soil		$10^3 - 10^4$		(Yuan et al., 2017)
Batch test	Sediment	Distilled water	3.59		(Yavar Ashayeri & Keshavarzi, 2019)
Batch method	Lake sediments	deionized water	3.4 - 4.5		(Li et al., 2017)
Batch experiment (solid-solution ratio: 1:12.5; contact time: 1 h; 25°C)	Aquifer sediment		5.08 - 17.3		(Guo et al., 2014)
Batch experiment (solid-solution ratio: 1:2; contact time: 72 h)	Soils		25 - 108	4 - 358 mg/kg	(Mrdakovic Popic et al., 2014)
Batch method	Original JL soil		3.89 - 31.62	7.5 - 14162.7 pmol/cm ³	(Wang et al., 2018)
	Stream bed sediment		$3 < K_d$		(Sedeño-Díaz et al., 2019)
Batch experiments in an open laboratory, 7 days at 25°C	Holocene aquifer sediments	Deionized water	30 - 39		(Chakraborty et al., 2014)
Batch experiment (solid-solution ratio: 0.5:10; contact time: 24 h; 25°C)	Sediment		1474 - 4255		(Cui et al., 2021)
Batch method	Sediment		130		(Desbarats et al., 2015)
Batch experiment (solid-solution ratio: 1:10; contact time: 24 h; 25°C)	Amended soil with HFO particles (first harvest)	Deionized water	$1.7 \times 10^3 - 5.4 \times 10^3$		(Huo et al., 2018)
	Amended soil with HFO particles (second harvest)		$1.9 \times 10^3 - 5.4 \times 10^3$		

Continued

Batch method	Sediment: Time treatment (0 - 80 day)		302 - 988	(Lin et al., 2017a)
Batch method	0-year aquaculture pond sediment	Milli-Q water	4.7 - 5.4	(Lin et al., 2017b)
	0-year aquaculture pond sediment		4.2 - 4.5	
	Cow Dung		4.7 - 6.3	
	Chicken Dung		5.1 - 6.0	
Batch experiment (solid-solution ratio: 1:4; contact time: 24 h; 23°C)	Marine EST port sediments	Water	55	(Mamindy-Pajany et al., 2013)
	Marine SM port sediments		450	
Batch experiment (solid-solution ratio: 2.5:24; contact time: 7 days)	Initial soil		4000 - 140,000	(Verbeeck et al., 2020)
	Aerobic soil		5200 - 160,000	
	Anaerobic soil		12 - 42,000	
	Anaerobic soil + glucose		12.0 - 7900	
Batch experiment	Sediments		6.50×10^1 - 2.93×10^4	(Boyer et al., 2018)
Total			27	19

1) Empty places in the tables are representing the not found data; 2) italic and bold values represent the log values of K_d collected; 3) bold and nonitalic values represent the mean values of K_d collected; 4) nonbold and nonitalic values represent the nonconverted values of K_d collected.

concluded that sediment could act as both a sink and a potential source of heavy metals in aquatic ecosystems. They added that high K_d values of Pb sorption demonstrate that the sediment preferentially retains the metal via adsorption reactions, which suggests the metal affinity and enrichment in sediment samples. Another study (Yavar Ashayeri & Keshavarzi, 2019) reported relatively high K_d values ($\log K_d = 4.23$ L/kg) found for Pb, suggesting its affinity and enrichment in sediment samples and indicating that Pb is distinguished by low geochemical mobility in water. Moreover, other authors suggest that Pb's high $\log K_d$ value could be due to its low solubility (Gu et al., 2014; Wokhe, 2015).

4.1.3. Copper (Cu)

Cu is an essential element for plants, animals, and micro-organisms, but it is toxic above a certain critical concentration. The sorption equilibrium that governs Cu exchange between solid and liquid phases determines its phytotoxicity

and the risk of contamination of water resources. The mobility and fate of Cu in the soil environment are directly related to its K_d , which indicates the ability of soil to retain a solute as well as the extent of its movement into the liquid phase (Christiansen et al., 2015; Ding et al., 2018; Kumar et al., 2019). Numerous studies have shown that the K_d values of Cu sorption are influenced by the variation of the added Cu concentration. It has been found that K_d values of Cu sorption decreased drastically as their added concentrations increased. In this direction, (Baghernejad et al., 2014) obtained K_d values of Cu decreasing from 3010 to 9.0 and 7495 to 42 L/kg in the Shekarbani and Sepidan soil series, respectively, for Cu added concentrations are increasing from 20 to 2000 mg/L. A similar variation of Cu K_d values was found in river sediments by (Fan et al., 2017), who assessed the concentration effect on Cu mobility in river sediments. The estimated K_d values of Cu decreased from 21.3 to 610, 39.4 to 413, 703 to 1.93×10^3 , 21.0 to 283, 25.4 to 147, and 86.6 to 919 L/kg for samples 1-S-L, 1-S-M, and 1-R-S sediments, respectively.

Furthermore, another author, Jalayeri et al. (2015) has also shown that the estimated K_d values of Cu absorption decreased from 5.31 to 0.6 and 3.41 to 0.55 L/kg for SA and SE soils, respectively, with an increase in initially added Cu concentration from 30 to 100 mg/L. The Same effect of Cu content has been observed by Alandis et al. (2019), who reported that K_d values of Cu occurred between 0 and 150 L/kg and decreased when Cu concentrations increased from 20 to 2200 mg/L. In contrast, Baghenejad et al. (2016) found that K_d values of Cu variations are not constant compared to those of other metals. So, the authors found that the K_d values of Cu increased from 2186 to 5179, 1116 to 2053, and 2512 to 5623 L/kg for soils 1, 3, and 4, with an increase of added Cu concentrations of 10 - 50 mg/L; and from 2659 to 4338, and 1675 to 1965 L/kg for soils 2 and 5, with an increase of added Cu concentrations of 10 - 20 mg/L. While Cu K_d values in soil 6 decreased from 3313 to 89 L/kg as their added concentrations increased from 10 - 200 mg/L, respectively, according to the authors, differences Cu sorption behaviors of studied soils almost certainly due to the differences in physical and chemical soil parameters, because Cu sorption is controlled by the soil organic matter (SOM) content, even in mineral soils.

In the literature, a strong relationship between K_d values of Cu and soil type, texture, and profile has been reported. In calcareous soil samples from south of Shiraz, Mahzari et al. (2013) found that the K_d value of Cu in calcareous soil from south of Shiraz, Iran, was 5517.3 L/kg. According to the researchers, the higher affinity for Cu in the studied soils is due to the presence of a more significant number of active sites (mostly organic matter) with high specificity for Cu, which means that when it is present, these sites are not occupied by other cations. Borah et al. (2018) used a batch adsorption method to examine the effect of sediment texture on assessing Cu distribution in tropical (Brahmaputra) river bed sediment in Assam, India. The K_d values of Cu determined varied from 0.98 to 1.16 L/kg. The authors reported that three factors, including textural drive,

have governed Cu enrichment and distribution. Huang et al. (2012) found that the $\log K_d$ values of Cu ranged from 3.58 to 5.41 L/kg. According to the authors, the higher K_d of Cu found at a slower velocity during the sediment resuspension could be attributed to the decrease of fine particles (silt/clay fraction) during resuspension. In two different soil textures of the surface layer, 0 - 0.3 m with clay loam and sandy loam (Al-Hassoon et al., 2019) carried out Cu adsorption experiments. The K_d values of Cu were found to vary between 8316 and 14476 L/kg. Yuan et al. (2017) studied the adsorption of Cu in soils flooded by smelting wastewater in Hechi, China. Cu K_d values in the soil decrease with profile and range from 103 to 104 L/kg. Zhang et al. (2018) used the batch method to assess the spatial distribution and correlation characteristics of Cu in the sediment-seawater system of Zhanjiang Bay, China. The estimated $\log K_d$ values ranged from 3.49 to 3.95 and 4.03 to 4.51 L/kg in January 2014 and June 2014, respectively, and from 3.49 to 4.51 L/kg in 2014. In a batch experiment conducted by Janik et al. (2015), the established Cu K_d values ranged from 1766.2 to 4317.9 L/kg for arable land (0 - 20 cm) and land under permanent grass cover (0 - 10 cm).

4.1.4. Zinc (Zn)

Zn is a trace element that is required for proper plant growth and reproduction, as well as animal and human health. However, when its concentration exceeds a critical value, it is considered a toxic element that can contaminate soil, water, and food chains (Noulas et al., 2018). Many studies have shown that the K_d values of Zn are influenced by both Zn properties (added Zn initial concentrations, and sorption systems) and the soil properties (such as soil type and texture, pH, clay content, organic matter, iron, and manganese oxides) (Das & Das, 2015; Gurpreet et al., 2012; Piri et al., 2019; Swati & Hait, 2017; Urbaniak et al., 2017; Vithanage et al., 2017). Generally, the K_d values of Zn decrease with the increase of added Zn initial concentrations (Azouzi et al., 2015). In this respect, Baghernejad et al. (2014) performed the batch method to study the concentration effect on adsorption of Zn in clay minerals of calcareous soils. The K_d values of Zn obtained decreased from 1275 to 6 and 5108 to 39 L/kg when added Zn concentrations increased from 20 to 2000 mg/L in the Shekarbani and Sepidan soil series, respectively. A similar variation of Zn was found by Baghernejad et al. (2016), who studied the adsorption of Zn in calcareous soils of southern Iran. The results showed that K_d values of Zn decreased significantly from 5435 to 10, 5337 to 18, 2270 to 3, 4235 to 11, 4980 to 2.7, and 491 to 2.7 with an increase in their added concentrations from 10 to 200 mg/L, respectively for different studied soils.

It has been demonstrated that K_d values of Zn are directly proportional to soil solution pH, clay content, organic matter, and temperature of the experiment (Borah et al., 2018). The effects of pH and experiment reaction temperature on K_d values of Zn sorption were studied in acid and alkaline soils (Kim, 2014). The estimated K_d means values of Zn ranged from 19.9 to 7739.3. The result shows

that the maximum K_d values of Zn adsorption are obtained at high pH. These results are similar to those observed by [Abat et al. \(2012\)](#) and [Li et al. \(2017\)](#) at high pH. The decrease in competition with H^+ for binding sites, the increase in the negative charge of the soil surface, and the increase in the proportion of hydrated ions. In contrast, in another study by [Azouzi et al. \(2015\)](#), the K_d values of Zn were found to decrease as pH increased. K_d value at pH = 6 were 1 - 3 times higher than at pH = 7.8. Furthermore, the result showed that increasing in temperature from $25^\circ C \pm 2^\circ C$ to $40^\circ C \pm 2^\circ C$ increased zinc uptake by 4.37% - 63.2% and 3.75% - 27.09% respectively at pH 7.8 and 6. However, the removal of organic matter slightly increased zinc sorption at alkaline pH while significantly decreasing it at acidic pH, indicating that the effect of organic matter was pH dependent.

Many studies have shown that the K_d values of Zn varied with the application of organic and inorganic amendments to soils ([Das & Das, 2015](#); [Mohseni et al., 2020](#); [Urbaniak et al., 2017](#); [Vithanage et al., 2017](#)). [Venegas et al. \(2015\)](#) evaluated the viability of compost from municipal organic waste, municipal solid waste, green waste derived from food leftovers, olive wet husk, olive pomace, and biochar derived from tree barks and vine shoots as amendments for the remediation of Zn contaminated soils. They found that green waste, tree barks, municipal organic waste, and vine shoots which have K_d values of Zn ranging from 80 to 1410, 105 to 515, 440 to 1220, and 85 to 2660 L/kg, respectively, are the best materials for environmental remediation that can be used alone or in mixtures to increase soil pH and sorption capacity. Biosolids were studied for their effects on the competitive sorption and lability of Zn in fluvial and calcareous soil ([Shaheen et al., 2017](#)). The reported K_d values ranged from 16.1 to 1334.9 L/kg for biosolids-amended fluvial soil and 7 to 490.2 L/kg for biosolids-amended calcareous soil. In another study, [Das & Das \(2015\)](#) investigated the influence of fly ash (FA) application on Zn adsorption-desorption in recommended chemical fertilizer (RDF) and farmyard manure (FYM) treatments of acidic Inceptisols of Assam. They found that the adsorption was most significant in the treatment receiving FA only at 15 t/ha and the least in the treatment receiving RDF 50% + FYM 5 t/ha + FA 5 t/ha. The estimated K_d values of treatment FA 15 t/ha ranged from 90.56 to 2.23 L/kg, which was 40 to 31 times higher than treatments containing FA + RDF + FYM. Furthermore, when FA was combined with RDF and FYM, Zn supply parameters increased, and Zn desorption occurred in the following order: $CaCl_2 > MgCl_2 > DTPA > HCl$. Finally, they concluded that the combination of fly ash, RDF, and FYM can effectively maintain significant Zn concentrations in soil.

4.1.5. Arsenic (As)

Arsenic is a naturally occurring trace element that is harmful to human and ecosystem health, especially when it is present in food or water supplies ([Al-Jumaily, 2016](#); [Rahman et al., 2020](#)). In the soil environment, As is found in two distinct chemical species: (1) arsenic as a hydroxyl species (H_3AsO_3 , $H_2AsO_3^-$) and (2)

arsenate as an oxyanion (H_2AsO_4^- or HAsO_4^{2-}) (Strawn, 2018; Zafeiriou et al., 2019). The risk of As to the environment is determined by the sorption that governs its exchange between solid and liquid phases (Almeida et al., 2021). Several factors, including pH, redox potential, minerals, organic matter, and cation exchange capacity (CEC), are well-known for influencing the K_d of As adsorption and desorption (Guo et al., 2014; Huo et al., 2018; Kader et al., 2016; Lee et al., 2020; Mamindy-Pajany et al., 2013; Mrdakovic Popic et al., 2014). Borah et al. (2018) studied the relationship between the K_d values of As and sediment texture, pH, CEC, organic content, and conductivity in river bed sediment in Assam, India. The K_d values determined varied from 1.06 to 1.74 L/kg. They found that the distribution of As was relatively higher on the downstream side due to the increase in pH, CEC, and clay content of the sediment. The same trend was observed for K_d values of As sorption by Chakraborty et al. (2014). They conducted As (III) adsorption studies in an open atmosphere at shallow aquifer sediments under oxidizing conditions. They found that the K_d values of As sorption varied from 30 to 39 L/kg over pH ranging from 6.0 to 9.1. However, in another study, Yavar Ashayeri & Keshavarzi (2019) carried out the batch method, no linear correlation was found between $\log K_d$ of As and pH, implying that As retention in sediments is not sensitive to pH fluctuations in the wetland. The reported mean value of $\log K_d$ of As reported was 3.59 L/kg. Kandakji et al. (2015) investigated As sorption characteristics and interactions with soil constituents in important agricultural soils using a batch sorption method. They found that the K_d for these semi-arid soils correlated negatively trend with pH (-0.81), sand (-0.95), and OM ($R = 0.93$, $n = 4$), Fe_{CBD} (0.88), clay (0.99), total Al (0.96), total Fe (0.97), and total Mn (0.98).

A strong relationship between organic amendments and K_d values of As has been reported in the literature. Lin et al. (2017b) measure and compare conditional K_d for As^{III} oxyanions with four different types of NOM from cow dung, chicken dung, and Bangladeshi aquaculture pond sediment before and after one year of operation. On the one hand, their results showed that As-sorption experiments with cow dung as the source of NOM resulted in the highest range for $\log K_d$, from 4.7 to 6.3 L/kg, compared to chicken dung with $\log K_d$ ranging from 5.1 to 6.0 L/kg. On the other hand, after a year of operation for fish production, pond sediment from Bangladesh showed a greater affinity for binding As oxyanions than fresh sediment before fish production. A batch sorption experiment was performed by Verbeeck et al. (2020) to study the role of soil organic matter (SOM) on the change in As mobility upon waterlogging soils. The reported K_d values of As sorption ranged from 4×10^3 to 1.4×10^5 L/kg for initial soil, 5.2×10^3 to 1.6×10^5 L/kg for aerobic soil, 12×10^0 to 4.2×10^4 L/kg for anaerobic soil, and 12×10^0 to 7.9×10^3 L/kg for anaerobic + glucose.

4.2. Factors Affecting K_d Values of Soil Heavy Metals: A Brief Comparative Description

The K_d represents the net result of various processes that transfer heavy metal

ions between the soil/sediment solid and solution. It plays a key role in predicting the fate and transport of heavy metals in the environment. Almost of afore-mentioned previous investigations demonstrate that K_d values of interest in heavy metals do not only depend on the chemical and physical characteristics of soil but also on the nature and characteristics of the metal involved as well as the experimental conditions (such as experiments time and temperature, and some added material to experiment soils) (Behbahani et al., 2020; Braz et al., 2013; Lin et al., 2017a; Rezaei et al., 2021; Shaheen et al., 2013).

4.2.1. Soil pH, CEC, Clay Content, Organic Matter, Iron Oxides, and Reaction Time

The literature has found a strong relationship between soil pH and heavy metal K_d values. Metal K_d values are directly proportional to soil pH (Kim, 2014; Shaheen et al., 2018; Tahervand & Jalali, 2017; Zhao et al., 2014). Metal element cations have been reported to be retained on soil surfaces as soil pH rises through sorption, inner-sphere surface complexation, and/or precipitation, as well as multinuclear type reactions. This can be explained by the fact that at low pH values, competition between cations and exuberance of H^+ ions for available permanent charged sites restricts the sorption of potentially toxic metals onto these sites, whereas at high pH values, this competition becomes feebler, and thus, more metal is adsorbed (Huang et al., 2014; Tahervand & Jalali, 2017). However, Yavar Ashayeri & Keshavarzi (2019) and Azouzi et al. (2015) have mentioned that no linear correlation was found between K_d and pH.

Several studies have shown that K_d also depends on the combination of pH, CEC, and clay content (Kader et al., 2016; Lee et al., 2020). In this respect, (Borah et al., 2018) found that the distribution of As was relatively higher on the downstream side due to an increase in pH, CEC, and clay content of the sediment. Soares et al. (2021) also reported that the K_d values of Cu in 30 soils of temperate regions (the State of Sao Paulo, southeastern Brazil) were influenced by the combined effect of effective cation exchange capacity (ECEC), contents of clay, and organic carbon. These results are similar to those reported by Aishah et al. (2018) who reported that the K_d values of Cu adsorption-desorption were higher in the Ultisol compared to that of the Oxisol, which was due to Ultisol's having higher CEC and OM content in comparison to that of the Oxisol. Clay and organic matters are dependent on K_d values of heavy metals because 1) the clay minerals (such as montmorillonite, imogolite, vermiculite, and amorphous allophanes reveal the highest sorption capacity) are a significant source of negative surface charges in soil and are a major contributor to their cation exchange capacity, particularly in mineral soils. The ability of clays to bind element ions is correlated with their CEC; usually, the greater the CEC, the greater the amount of cation adsorbed. 2) In the case of organic matter, the CEC is high due to the dissociation of organic acids present on the surface and other functional groups.

Like soil organic matters, the presence of iron oxides in soil sand sediment soils and have a significant impact on the transfer, transformation, and immobi-

lization of heavy metals in soils (Roth et al., 2012). They possess high molecular weights, exhibit low mobility in soil, and have various functional groups, endowing them with a high capacity to immobilize potentially toxic metals. The metal distribution and redistribution patterns for untreated and treated soil showed that soils with high organic matter content retained more metal ions than those with high iron oxide content in the short term (Diagboya et al., 2015). While in the long term, however, high organic matter content led to reduced metal retention and increased desorption with time, while iron oxides enhanced retention and retarded desorption with time. As a result, soil organic matter was important in the short-term sorption of metals, while iron oxides were important at longer times.

4.2.2. Metal Concentration, Type, Sorption Systems, Biosolids, and Wastewater

Previous research has shown that the initial heavy metal concentration in the experiment solution can influence the K_d at both low and high concentrations. In general, the K_d values decrease as the concentration of the included metal cation in the experiment solution increases (Baghernejad et al., 2016). Several previous studies (Baghernejad et al., 2014; Fan et al., 2017) found that heavy metal concentration in the experiment solution had an effect on K_d values in porous media. Cd sorption K_d values were found to be high at low initial concentrations and increased with an increase in the initial concentration (Rezaei et al., 2021), in contrast to high concentrations. According to the literature, higher K_d values obtained with lower metal concentrations are associated with high selectivity sorption sites with relatively strong binding energies. Otherwise, metal sorption becomes unspecific at higher element concentrations as the specific binding sites become increasingly occupied, resulting in lower K_d values. In other words, at low heavy metal concentrations, they are primarily adsorbed onto specific sorption sites, whereas at higher element concentrations, soils lose some of their ability to bind trace elements as sorption sites overlap, making a particular element less specific.

Previous research has shown that the variation of K_d values of heavy metals depends on the type, properties, and nature of the elements involved. However, soil sorption preference for one metal over others may be due to the following factors: 1) the hydrolysis constant; 2) the atomic weight; 3) the ionic radius and, later, hydrated radius; and 4) its Misono softness value (Shaheen et al., 2013). This is usually attributed to differences in heavy metal properties and the resulting affinity for sorption sites. For example, in the case of Pb, the affinity for sorption sites is because the hydrated radius (0.401 nm) of Pb^{2+} is smaller than that of Cd^{2+} (0.426 nm) and favors Coulombic interactions of Pb with exchange sites. Furthermore, Pb has a greater affinity for most functional groups in organic matter, including carboxylic and phenolic groups, which are hard Lewis bases. This is mainly attributed to the differences in the chemical properties of the two elements (Pb and Cd).

It is also found that when heavy metal CS is compared to their MS behavior, their sorption in CS is lower (Shaheen et al., 2015). When CS K_d values of Pb are compared to their MS behavior, it has been reported that CS has lower K_d values. Similarly, Li et al. (2012) used batch equilibrium experiments to investigate the sorption and desorption of Pb on paddy soils using MS and CS systems. Their findings showed that K_d values of Pb in MS were higher than K_d of Pb in CS. These findings are consistent with those reported by Mahzari et al. (2013) who also found that K_d values of Pb in MS were higher than K_d of Pb in CS. It can be seen that CS has a suppressive effect on the sorption of the metal element, indicating that the metal element is less retained in the soil under CS.

Furthermore, several studies have shown that the sorption systems are affected by the type of soil and metal elements involved (Li et al., 2017). Oladipupo Azeez et al. (2018) found that under the MS experiment, Pb had the highest K_d value in both acid and alkaline soils, Zn had the highest K_d value in slightly acid soil, and Cu had the lowest K_d value across the three soil types. While in the CS experiment, Zn and Cd had the highest and lowest K_d value, respectively, across the three soil types. According to the authors, this may be due to the lower ionic radius of Cu^{2+} compared to Cd^{2+} (0.72 Å versus 0.97 Å), thus Cu^{2+} can conveniently enter into the soil interlayer, suggesting the greater exchangeable adsorption rate of Cu^{2+} . This result agrees with previous studies on heavy metal sorption systems (Shaheen et al., 2012), which found that Pb, the most strongly sorbed metal in most cases of study, was less affected by competition than other metals. While Zn and Cd, the most poorly sorbed metals compared to Pb, were greatly affected by competition.

Several attempts have been made to investigate the effects of soil amendments to mobilize/immobilize metal (Kaninga et al., 2020; Lin et al., 2017b; Sharma et al., 2017; Verbeeck et al., 2020). The sewage sludge effect on Zn desorption was studied by Mohseni et al. (2020). They found that the K_d values of Zn desorption increased in soils treated with high (30 g/kg) sewage sludge, resulting in a significant increase in plant Zn concentrations. While in soils treated with a small quantity of sewage sludge (10 g/kg), the K_d values of Zn desorption decreased over incubation time. Shaheen et al. (2018) also reported similar results and found that the K_d values of Pb sorption increased in the Alfisol at high biosolid application rates. The authors suggested that this might immobilize Pb and decrease its solubility.

In contrast, the application of a low biosolids rate to Entisol decreased the K_d values of Pb sorption and thus might mobilize Pb and increase its solubility and enhance its phytoextraction. Other authors, Al-Oud & Ghoneim (2018) investigate the effects of municipal solid waste ash (MSWA) application rates on the mobility of Pb in 2 soils with different properties. The results indicated that K_d values of Pb on sandy loam soil were higher than those on sandy loam soil. Whereas, they found that an application rate of 5% MSWA to loamy sand and sandy loam soils resulted in increases of K_d values of 36.6% and 29.0% more than the control soil (0%). They conclude that the MSWA amendment is most effec-

tive in reducing Pb mobility in the studied soils.

4.3. Characteristics of the Number of K_d Data Observations

Table 6 shows the total numbers of the observation of references and K_d data points; and the variation of K_d data points in sorption systems (MS and CS) for the five heavy metals. On the one hand, regarding the variation of the K_d data points in sorption systems, it can be seen that the reported K_d values showed a wide variability of magnitude: 10^{-2} to 10^6 ; for Cd, As, and Zn, 10^{-2} to 10^7 for Pb; and 10^{-2} to 10^5 for Cu, respectively for studies performed in MS. While on CS, K_d values ranged from 10^{-2} to 10^4 for Cd, Pb, and Zn, and from 10^{-2} to 10^5 for Cu, respectively. The K_d values for arsenic in CS have not been determined. This wide range of K_d variability might have resulted from different environmental conditions such as experimental methods, sorbing materials, metal characteristics, equilibration time, etc. These findings, on the other hand, revealed that the K_d values of heavy metals decreased in CS when compared to MS. This was consistent with previous findings that competitive sorption of metals was lower in competitive systems than in mono-metal systems (Baghenejad et al., 2016; Oladipupo Azeez et al., 2018; Shaheen et al., 2012; Shaheen et al., 2013; Shaheen et al., 2015). A surface-active site can sorb different ions, but once an ion has been adsorbed, no other ions can be adsorbed at the same active site. Our findings completely confirm this effect of the sorption systems on K_d . In addition, the total number of references observed in **Table 6**, showed that about half of reviewer articles were done on Cd 35 (50%) and Zn 35 (50%); less than half were done on Pb 33 (47.14%) and Cu 33 (47.14%), and less than two-sevenths were made on As 19 (27.14%).

Table 7 and **Figure 2** present the reference numbers of K_d data observation in sorption systems, conditions of liquid-solid exchange, and environmental components for the five heavy metals. Our findings showed that the majority of research articles reviewed were conducted: 1) in MS rather than CS for sorption systems, 2) in sorption rather than desorption conditions for liquid-solid exchange conditions, and 3) in soil rather than sediments for environmental components (**Table 7** and **Figure 2**). Although adsorption and desorption have been

Table 6. K_d data in the function of sorption systems.

Element	Number of references.	Number of data	sorption system	
			MS	CS
Cd	34	64	10^{-2} - 10^6	10^{-2} - 10^4
Pb	33	56	10^{-2} - 10^7	10^{-2} - 10^4
Cu	33	58	10^{-2} - 10^5	10^{-2} - 10^5
Zn	34	72	10^{-2} - 10^6	10^{-2} - 10^4
As	19	27	10^{-2} - 10^6	None

Table 7. K_d data number (value of K_d has appeared for an element n times) in the function of references, data, sorption system, conditions of liquid-solid exchange, and environmental components.

Element	Number of references.	Number of data	Sorption system			Conditions of liquid-solid exchange			Environmental components	
			MS	CS	MS-CS	Sorption	Desorption	Sorption-desorption	Soil	Sediment
Cd	34	64	25	5	4	25	2	7	25	9
Pb	33	56	26	4	3	24	1	8	22	11
Cu	33	58	26	6	1	22	1	10	24	9
Zn	34	72	29	3	2	21	2	11	24	10
As	19	27	19	0	0	12	0	7	7	12
Total:		277	125	18	10	104	6	43	102	51

identified as the most important mechanisms which control metal ion bioavailability, transport, and transformation in soil and sediment (Aishah et al., 2018), most articles observed in this review have been carried out on metal sorption. A similar observation was made by Jiang et al. (2012). They reported that previous research focused primarily on the adsorption process, with much less information on the desorption process. Because the solubility and bioavailability of heavy metals in soil vary significantly, depending on the nature of both adsorption and desorption processes, studying both adsorption and desorption processes simultaneously in the same experiment conditions may lead to a better understanding of metal bioavailability, phytotoxicity, and ultimate fate in the environment (Sparks, 2003).

5. Conclusions and Prospects

This paper examined the K_d values of five heavy metals. Furthermore, we present various methods for estimating K_d values and provide subsequent compilations of K_d data on Cd, Pb, Cu, Zn, and As in soil/sediment under various aquatic mediums. We found that the K_d values of heavy metals are affected by various factors, including MS, CS, element metal properties, physical and chemical properties of soil, and experimental conditions. The K_d values were almost higher at low concentrations and decreased with the increase of metal concentrations. Unlike the metal concentration, the K_d of heavy metals increases with the increase in pH value, so it is higher in calcareous soils than in acidic soils. Through the literature, we discovered that K_d values in both organic matter removal and Mn oxide removal soils were lower when compared to untreated soils.

Furthermore, the reported K_d values of heavy metals showed a wide range of magnitude variation, as follows: in MS, Pb was 10^{-2} to 10^7 , Cd, As, and Zn were 10^{-2} to 10^6 , Cu was 10^{-2} to 10^5 ; in CS, K_d values ranged from 10^{-2} to 10^5 for Cu, and from 10^{-2} to 10^4 for Pb, Cd, and Zn, respectively. Values of K_d for As in CS have not been determined. Heavy metals with the highest K_d values are relatively insoluble and migrate slowly. With regard to the numbers of references, the highest numbers of references were found for Cd and Zn followed by Pb and Cu,

and the lowest for As. These results showed that over the last decade, most of the reviewed studies conducted to investigate the K_d values were focused mainly on the Cd and Zn contamination, followed by Pb and Cu, with much less information available for As. In addition, our findings also showed that the majority of research was conducted: 1) more on MS than on CS for sorption systems, 2) more on sorption than on desorption for liquid-solid exchange conditions, 3) more on soil than on sediments for environmental components, and 4) most literature have reported the K_d values, derived from batch method than on column method.

In general, several studies have been conducted to investigate the role of heavy metal sorption and desorption K_d values in assessing their mobility in soil. But, the experimental conditions, soil physical and chemical properties, and metal properties have been fairly diverse, making it hard to compare the results to identify general trends and draw conclusions. Thus, due to the complexity of the process, additional investigations are still critical. 1) To deeply understand the practical utilization of studying K_d values of heavy metals for assessing the remediation process of metals in soil, a critical review of recent existing literature concerning K_d values is needed not only to summarize and compare the obtained results and conclusions but also to be able to deduce some standards K_d values or magnitudes to be used for each method. 2) More studies regarding heavy metal transport and adsorption-desorption in soils under the same experimental conditions are needed, precisely through column experimental investigation.

Acknowledgements

This work was supported by the National Key R&D Program of China (2019-YFC1804400) and the National Natural Scientific Foundation of China (42171036). The authors would like to acknowledge the financial support from Tianjin University.

Conflicts of Interest

The authors declare no conflicts of interest regarding the publication of this paper.

References

- Abat, M., McLaughlin, M. J., Kirby, J. K., & Stacey, S. P. (2012). Adsorption and Desorption of Copper and Zinc in Tropical Peat Soils of Sarawak, Malaysia. *Geoderma*, 175-176, 58-63. <https://doi.org/10.1016/j.geoderma.2012.01.024>
- Aishah, R. M., Shamshuddin, J., Fauziah, C. I., Abdu Arifin, A., & Panhwar, Q. A. (2018). Adsorption-Desorption Characteristics of Zinc and Copper in Oxisol and Ultisol Amended with Sewage Sludge. *Journal of the Chemical Society of Pakistan*, 40, 842-855.
- Alandis, N. M., Mekhamer, W., Aldayel, O., Hefne, J. A. A., & Alam, M. (2019). Adsorptive Applications of Montmorillonite Clay for the Removal of Ag(I) and Cu(II) from Aqueous Medium. *Journal of Chemistry*, 2019, Article ID: 7129014.

- <https://doi.org/10.1155/2019/7129014>
- Al-Hassoon, S. N. H., Al-Hayani, A. S. J. Z., & Al-Obaidi, M. A. J. (2019). Copper Adsorption in Two Different Soil Texture. *Iraqi Journal of Agricultural Sciences*, *50*, 638-645.
- Al-Hayani, A. S. J. Z., & Al-Obaidi, M. A. J. (2019). Diffusion and Distribution Coefficient of Lead in Some Calcareous Soils in Northern Iraq. *Iraqi Journal of Soil Science*, *19*, 223-234.
- Al-Jumaily, H. A. A. (2016). An Evaluation Performance of Potential Pollution of Arsenic, Chromium and Cadmium in the Road Side Soil of Kirkuk City, Northern Iraq. *Journal of Geoscience and Environment Protection*, *4*, 80-94.
<https://doi.org/10.4236/gep.2016.49007>
- Allison, J. D., & Allison, T. L. (2005). *Partition Coefficients for Metals in Surface Water, Soil, and Waste*. U.S. Environmental Protection Agency, Office of Research and Development.
- Alloway, B. (2013). *Heavy Metals in Soils. Trace Metals and Metalloids in Soils and Their Bioavailability*. Springer. <https://doi.org/10.1007/978-94-007-4470-7>
- Almeida, C. C., Fontes, M. P. F., Dias, A. C., Pereira, T. T. C., & Ker, J. C. (2021). Adsorption and Desorption of Arsenic and Its Immobilization in Soils. *Scientia Agricola*, *78*, e20180386. <https://doi.org/10.1590/1678-992x-2018-0368>
- Al-Obaidi, M. A. J., & Al-Obadi, A. M. A. M. (2019). Lead Reactions in Some Calcareous Soils of Nineveh Governorate. *Mesopotamia Journal of Agriculture*, *47*, 8-26.
<https://doi.org/10.33899/magrj.2020.126310.1015>
- Al-Oud, S., & Ghoneim, A. (2018). Lead Transport in Soils Amended with Municipal Solid Waste Ash. *Polish Journal of Environmental Studies*, *27*, 2437-2442.
<https://doi.org/10.15244/pjoes/76030>
- Antoniadis, V., Shaheen, S. M., Tsadilas, C. D., Selim, M. H., & Rinklebe, J. (2018). Zinc Sorption by Different Soils as Affected by Selective Removal of Carbonates and Hydrrous Oxides. *Applied Geochemistry*, *88*, 49-58.
<https://doi.org/10.1016/j.apgeochem.2017.04.007>
- Azouzi, R., Charef, A., & Hamzaoui, A. H. (2015). Assessment of Effect of pH, Temperature and Organic Matter on Zinc Mobility in a Hydromorphic Soil. *Environmental Earth Sciences*, *74*, 2967-2980. <https://doi.org/10.1007/s12665-015-4328-4>
- Baghenejad, M., Javaheri, F., & Moosavi, A. A. (2016). Adsorption Isotherms of Some Heavy Metals under Conditions of their Competitive Adsorption onto Highly Calcareous Soils of Southern Iran. *Archives of Agronomy and Soil Science*, *62*, 1462-1473.
<https://doi.org/10.1080/03650340.2016.1147647>
- Baghernejad, M., Javaheri, F., & Moosavi, A. A. (2014). Adsorption Isotherms of Copper and Zinc in Clay Minerals of Calcareous Soils and Their Effects on X-Ray Diffraction. *Archives of Agronomy and Soil Science*, *61*, 1061-1077.
<https://doi.org/10.1080/03650340.2014.982549>
- Behbahani, A., Ryan, R. J., & McKenzie, E. R. (2020). Long-Term Simulation of Potentially Toxic Elements (PTEs) Accumulation and Breakthrough in Infiltration-Based Stormwater Management Practices (SMPs). *Journal of Contaminant Hydrology*, *234*, Article ID: 103685. <https://doi.org/10.1016/j.jconhyd.2020.103685>
- Bielska, L., Hovorkova, I., Kuta, J., Machat, J., & Hofman, J. (2017). The Variability of Standard Artificial Soils: Cadmium and Phenanthrene Sorption Measured by a Batch Equilibrium Method. *Ecotoxicology and Environmental Safety*, *135*, 17-23.
<https://doi.org/10.1016/j.ecoenv.2016.09.015>
- Borah, R., Taki, K., Gogoi, A., Das, P., & Kumar, M. (2018). Contemporary Distribution

- and Impending Mobility of Arsenic, Copper and Zinc in a Tropical (Brahmaputra) River Bed Sediments, Assam, India. *Ecotoxicology and Environmental Safety*, 161, 769-776. <https://doi.org/10.1016/j.ecoenv.2018.06.038>
- Boyer, P., Wells, C., & Howard, B. (2018). Extended Kd Distributions for Freshwater Environment. *Journal of Environmental Radioactivity*, 192, 128-142. <https://doi.org/10.1016/j.jenvrad.2018.06.006>
- Braz, A. M., Fernandes, A. R., Ferreira, J. R., & Alleoni, L. R. (2013). Prediction of the Distribution Coefficients of Metals in Amazonian Soils. *Ecotoxicology and Environmental Safety*, 95, 212-220. <https://doi.org/10.1016/j.ecoenv.2013.05.007>
- Chakraborty, S., Nath, B., Chatterjee, D., & Charlet, L. (2014). Retardation of Arsenic Transport by Oxidized Holocene Aquifer Sediments of West Bengal, India. *Journal of Hydrology*, 518, 460-463. <https://doi.org/10.1016/j.jhydrol.2013.07.028>
- Christiansen, K. S., Borggaard, O. K., Holm, P. E., Vijver, M. G., Hauschild, M. Z., & Peijnenburg, W. J. (2015). Experimental Determinations of Soil Copper Toxicity to Lettuce (*Lactuca sativa*) Growth in Highly Different Copper Spiked and Aged Soils. *Environmental Science and Pollution Research*, 22, 5283-5292. <https://doi.org/10.1007/s11356-014-3737-3>
- Cui, J., Wang, D., Lin, J., Wang, Y., Ren, M., Yang, Y., & Shi, P. (2021). New Application of Lanthanum-Modified Bentonite (Phoslock) for Immobilization of Arsenic in Sediments. *Environmental Science and Pollution Research*, 28, 2052-2062. <https://doi.org/10.1007/s11356-020-10565-x>
- Das, B. K., & Das, K. N. (2015). Adsorption-Desorption of Zinc as Influenced by Fly Ash in Inceptisols of Assam. *Communications in Soil Science and Plant Analysis*, 46, 800-818. <https://doi.org/10.1080/00103624.2015.1011748>
- de Oliveira Barros, V. H., Coutinho, A. P., Alves, A. T. A., dos Santos Neto, S. M., de Assis, F. X., de Lima, V. F., de Sousa Lima, J. R., & Antonino, A. C. D. (2021). Sorption Studies of Cd and Cu on Brazilian Alluvial Soils. *Ambiente e Agua—An Interdisciplinary Journal of Applied Science*, 16, e2717. <https://doi.org/10.4136/ambi-agua.2717>
- Desbarats, A. J., Parsons, M. B., & Percival, J. B. (2015). Arsenic Mobility in Mildly Alkaline Drainage from an Orogenic Lode Gold Deposit, Bralorne Mine, British Columbia. *Applied Geochemistry*, 57, 45-54. <https://doi.org/10.1016/j.apgeochem.2014.11.015>
- Diagboya, P. N., Olu-Owolabi, B. I., & Adebowale, K. O. (2015). Effects of Time, Soil Organic Matter, and Iron Oxides on the Relative Retention and Redistribution of Lead, Cadmium, and Copper on Soils. *Environmental Science and Pollution Research*, 22, 10331-10339. <https://doi.org/10.1007/s11356-015-4241-0>
- Ding, X., Ma, W., Huang, X., Guo, W., Wang, R., Li, Y., & Dai, J. (2018). Characteristics of Copper Sorption by Various Agricultural Soils in China and the Effect of Exogenic Dissolved Organic Matter on the Sorption. *Soil and Sediment Contamination: An International Journal*, 27, 311-328. <https://doi.org/10.1080/15320383.2018.1473333>
- Ea, A., & Grunzke, D. (2016). Multi-Regression Prediction of Metal Partition Coefficients under Various Physical/Chemical Conditions “Design of Experiments As, Cr, Cu, Ni and Zn”. *Journal of Waste Water Treatment & Analysis*, 7, Article ID: 1000241. <https://doi.org/10.4172/2157-7587.1000241>
- Egbi, C. D., Akiti, T. T., Osae, S., Dampare, S. B., Abass, G., & Adomako, D. (2015). Assessment of Groundwater Quality by Unsaturated Zone Study due to Migration of Leachate from Abloradjei Waste Disposal Site, Ghana. *Applied Water Science*, 7, 845-859. <https://doi.org/10.1007/s13201-015-0297-8>
- Elbana, T. A., Magdi Selim, H., Akrami, N., Newman, A., Shaheen, S. M., & Rinklebe, J. (2018). Freundlich Sorption Parameters for Cadmium, Copper, Nickel, Lead, and Linc

- for Different Soils: Influence of Kinetics. *Geoderma*, 324, 80-88. <https://doi.org/10.1016/j.geoderma.2018.03.019>
- EPA (1999). *Understanding Variation in Partition Coefficient, K_d, Values* (Vol. I).
- EPA (2004). *Understanding Variation in Partition Coefficient, K_d, Values* (Vol. III).
- Fan, J., Zhao, G., & Sun, J. (2017). Binary Component Sorption of Cadmium, and Copper Ions onto Yangtze River Sediments with Different Particle Sizes. *Sustainability*, 9, Article 2089. <https://doi.org/10.3390/su9112089>
- Gu, X., Sun, J., & Evans, L. J. (2014). The Development of a Multi-Surface Soil Speciation Model for Cd(II) and Pb(II): Comparison of Two Approaches for Metal Adsorption to Clay Fractions. *Applied Geochemistry*, 47, 99-108. <https://doi.org/10.1016/j.apgeochem.2014.05.014>
- Guo, F., Ding, C., Zhou, Z., Huang, G., & Wang, X. (2018). Effects of Combined Amendments on Crop Yield and Cadmium Uptake in Two Cadmium Contaminated Soils under Rice-Wheat Rotation. *Ecotoxicology and Environmental Safety*, 148, 303-310. <https://doi.org/10.1016/j.ecoenv.2017.10.043>
- Guo, G., Zhang, Y., Zhang, C., Wang, S., Yan, Z., & Li, F. (2013). Partition and Characterization of Cadmium on Different Particle-Size Aggregates in Chinese Phaeozem. *Geoderma*, 200-201, 108-113. <https://doi.org/10.1016/j.geoderma.2013.01.013>
- Guo, Q., Guo, H., Yang, Y., Han, S., & Zhang, F. (2014). Hydrogeochemical Contrasts between Low and High Arsenic Groundwater and Its Implications for Arsenic Mobilization in Shallow Aquifers of the Northern Yinchuan Basin, P.R. China. *Journal of Hydrology*, 518, 464-476. <https://doi.org/10.1016/j.jhydrol.2014.06.026>
- Gurpreet, K., Sharma, B. D., & Sharma, P. (2012). Zinc Adsorption as Affected by Concentration, Temperature, and Time of Contact in the Presence of Electrolytic and Aqueous Medium in Benchmark Soils of Punjab in Northwest India. *Communications in Soil Science and Plant Analysis*, 43, 701-715. <https://doi.org/10.1080/00103624.2012.644009>
- Huang, B., Li, Z., Huang, J., Guo, L., Nie, X., Wang, Y., Zhang, Y., & Zeng, G. (2014). Adsorption Characteristics of Cu and Zn onto Various Size Fractions of Aggregates from Red Paddy Soil. *Journal of Hazardous Materials*, 264, 176-183. <https://doi.org/10.1016/j.jhazmat.2013.10.074>
- Huang, B., Yuan, Z., Li, D., Zheng, M., Nie, X., & Liao, Y. (2020). Effects of Soil Particle Size on the Adsorption, Distribution, and Migration Behaviors of Heavy Metal(loid)s in Soil: A Review. *Environmental Sciences: Processes and Impacts*, 22, 1596-1615. <https://doi.org/10.1039/D0EM00189A>
- Huang, J., Ge, X., Yang, X., Zheng, B., & Wang, D. (2012). Remobilization of Heavy Metals during the Resuspension of Liangshui River Sediments Using an Annular Flume. *Chinese Science Bulletin*, 57, 3567-3572. <https://doi.org/10.1007/s11434-012-5370-1>
- Huo, L., Huang, D., Zeng, X., Su, S., Wang, Y., Bai, L., & Wu, C. (2018). Arsenic Availability and Uptake by Edible Rape (*Brassica campestris* L.) Grown in Contaminated Soils Spiked with Carboxymethyl Cellulose-Stabilized Ferrihydrite Nanoparticles. *Environmental Science and Pollution Research*, 25, 15080-15088. <https://doi.org/10.1007/s11356-018-1718-7>
- Jaishankar, M., Tseten, T., Anbalagan, N., Mathew, B. B., & Beeregowda, K. N. (2014). Toxicity, Mechanism and Health Effects of Some Heavy Metals. *Interdisciplinary Toxicology*, 7, 60-72. <https://doi.org/10.2478/intox-2014-0009>
- Jakomin, L. M., Marban, L., Grondona, S., Glok Galli, M., & Martinez, D. E. (2015). Mobility of Heavy Metals (Pb, Cd, Zn) in the Pampeano and Puelche Aquifers, Argentina: Partition and Retardation Coefficients. *Bulletin of Environmental Contamination and Toxicology*, 95, 325-331. <https://doi.org/10.1007/s00128-015-1602-9>

- Jalali, M., & Hoursersht, Z. (2017). Metal Extractability in Binary and Multi-Metals Spiked Calcareous Soils. *Communications in Soil Science and Plant Analysis*, 48, 1089-1104. <https://doi.org/10.1080/00103624.2017.1341912>
- Jalali, M., & Vafaei, Z. (2017). The Effect of Electrolyte Type and Concentration on the Release of Cd, Cu, Ni, and Zn in Some Contaminated Calcareous Soils. *Soil and Sediment Contamination: An International Journal*, 26, 651-661. <https://doi.org/10.1080/15320383.2017.1397595>
- Jalayeri, H., Salarirad, M. M., & Ziiai, M. (2015). Behavior and Mechanism of Various Components of Soil in Cu(II) Adsorption from Aqueous Solution. *Desalination and Water Treatment*, 57, 8494-8503. <https://doi.org/10.1080/19443994.2015.1021853>
- Janik, L. J., Forrester, S. T., Soriano-Disla, J. M., Kirby, J. K., McLaughlin, M. J., Reimann, C., & Team, G. P. (2015). GEMAS: Prediction of Solid-Solution Partitioning Coefficients (Kd) for Cationic Metals in Soils Using Mid-Infrared Diffuse Reflectance Spectroscopy. *Environmental Toxicology and Chemistry*, 34, 224-234. <https://doi.org/10.1002/etc.2736>
- Jiang, H., Li, T., Han, X., Yang, X., & He, Z. (2012). Effects of pH and Low Molecular Weight Organic Acids on Competitive Adsorption and Desorption of Cadmium and Lead in Paddy Soils. *Environmental Monitoring and Assessment*, 184, 6325-6335. <https://doi.org/10.1007/s10661-011-2422-y>
- Kader, M., Lamb, D. T., Megharaj, M., & Naidu, R. (2016). Sorption Parameters as a Predictor of Arsenic Phytotoxicity in Australian Soils. *Geoderma*, 265, 103-110. <https://doi.org/10.1016/j.geoderma.2015.11.019>
- Kandakji, T., Udeigwe, T. K., Athanasiou, D., & Pappas, S. (2015). Chemistry of Arsenic in Semi-Arid Alkaline Soils of the Southern High Plains, USA: Sorption Characteristics and Interactions with Soil Constituents. *Water, Air, & Soil Pollution*, 226, Article No. 320. <https://doi.org/10.1007/s11270-015-2543-y>
- Kaninga, B., Chishala, B. H., Maseka, K. K., Sakala, G. M., Young, S. D., Lark, R. M., Tye, A., Hamilton, E. M., Gardner, A., & Watts, M. J. (2020). Do Soil Amendments Used to Improve Agricultural Productivity Have Consequences for Soils Contaminated with Heavy Metals? *Heliyon*, 6, e05502. <https://doi.org/10.1016/j.heliyon.2020.e05502>
- Khan, A., Khan, S., Khan, M. A., Qamar, Z., & Waqas, M. (2015). The Uptake and Bioaccumulation of Heavy Metals by Food Plants, their Effects on Plants Nutrients, and Associated Health Risk: A Review. *Environmental Science and Pollution Research*, 22, 13772-13799. <https://doi.org/10.1007/s11356-015-4881-0>
- Kim, C. S., Bai, B. H., Kim, P. B., & Chon, K. (2018). Review of Reviews: A Systematic Analysis of Review Papers in the Hospitality and Tourism Literature. *International Journal of Hospitality Management*, 70, 49-58. <https://doi.org/10.1016/j.ijhm.2017.10.023>
- Kim, M.-J. (2014). A Study on the Adsorption Characteristics of Cadmium and Zinc onto Acidic and Alkaline Soils. *Environmental Earth Sciences*, 72, 3981-3990. <https://doi.org/10.1007/s12665-014-3287-5>
- Kumar, A., Rout, S., Pulhani, V., & Kumar, A. V. (2019). A review on Distribution Coefficient (Kd) of Some Selected Radionuclides in Soil/Sediment over the Last Three Decades. *Journal of Radioanalytical and Nuclear Chemistry*, 323, 13-26. <https://doi.org/10.1007/s10967-019-06930-x>
- Lee, M., Ahn, Y., Pandi, K., Jo, H. Y., & Choi, J. (2020). Sorption of Bioavailable Arsenic on Clay and Iron Oxides Elevates the Soil Microbial Activity. *Water, Air, & Soil Pollution*, 231, Article No. 411. <https://doi.org/10.1007/s11270-020-04784-8>
- Li, T., Jiang, H., Yang, X., & He, Z. (2012). Competitive Sorption and Desorption of Cadmium and Lead in Paddy Soils of Eastern China. *Environmental Earth Sciences*, 68,

- 1599-1607. <https://doi.org/10.1007/s12665-012-1853-2>
- Li, Y., Liu, F., Zhou, X., Wang, X., Liu, Q., Zhu, P., Zhang, L., & Sun, C. (2017). Distribution and Ecological Risk Assessment of Heavy Metals in Sediments in Chinese Collapsed Lakes. *Polish Journal of Environmental Studies*, 26, 181-188. <https://doi.org/10.15244/pjoes/64379>
- Liao, J., Qian, X., Liu, F., Deng, S., Lin, H., Liu, X., & Wei, C. (2021). Multiphase Distribution and Migration Characteristics of Heavy Metals in Typical Sandy Intertidal Zones: Insights from Solid-liquid Partitioning. *Ecotoxicology and Environmental Safety*, 208, Article ID: 111674. <https://doi.org/10.1016/j.ecoenv.2020.111674>
- Lin, J., Sun, Q., Ding, S., Wang, D., Wang, Y., & Tsang, D. C. W. (2017a). First Observation of Labile Arsenic Stratification in Aluminum Sulfate-Amended Sediments Using High Resolution Zr-Oxide DGT. *Science of the Total Environment*, 609, 304-310. <https://doi.org/10.1016/j.scitotenv.2017.07.165>
- Lin, T. Y., Hafeznezami, S., Rice, L., Lee, J., Maki, A., Sevilla, T., Stahl, M., Neumann, R., Harvey, C., Suffet, I. H., Badruzzaman, A. B. M., & Jay, J. A. (2017b). Arsenic Oxyanion Binding to NOM from Dung and Aquaculture Pond Sediments in Bangladesh: Importance of Site-Specific Binding Constants. *Applied Geochemistry*, 78, 234-240. <https://doi.org/10.1016/j.apgeochem.2016.12.026>
- Liu, S., Wang, Z., Zhang, Y., Liu, Y., Yuan, W., Zhang, T., Liu, Y., Li, P., He, L., & Chen, J. (2019). Distribution and Partitioning of Heavy Metals in Large Anthropogenically Impacted River, the Pearl River, China. *Acta Geochimica*, 38, 216-231. <https://doi.org/10.1007/s11631-018-00309-7>
- Loganathan, P., Vigneswaran, S., Kandasamy, J., & Naidu, R. (2012). Cadmium Sorption and Desorption in Soils: A Review. *Critical Reviews in Environmental Science and Technology*, 42, 489-533. <https://doi.org/10.1080/10643389.2010.520234>
- Mahzari, M., Siroienejad, A., & Moalem, M. (2013). Effect of Mn and Cu on the Adsorption of Lead in Soil Sample from South of Shiraz, Iran. *Asian Journal of Chemistry*, 25, 2389-2391. <https://doi.org/10.14233/ajchem.2013.13328>
- Maity, S., & Pandit, G. G. (2014). Estimation of Kd of Lead and (210)Po in 11 Soils from India. *Journal of Environmental Radioactivity*, 138, 434-437. <https://doi.org/10.1016/j.jenvrad.2014.03.021>
- Mamindy-Pajany, Y., Hurel, C., Geret, F., Galgani, F., Battaglia-Brunet, F., Marmier, N., & Romeo, M. (2013). Arsenic in Marine Sediments from French Mediterranean Ports: Geochemical Partitioning, Bioavailability and Ecotoxicology. *Chemosphere*, 90, 2730-2736. <https://doi.org/10.1016/j.chemosphere.2012.11.056>
- Mohseni, A., Reyhanitabar, A., Najafi, N., Oustan, S., & Bazargan, K. (2020). Effects of Sludge on Heavy Metals Release from Peppermint-Planted Soils during Time as Assessed by DGT Technique. *Archives of Agronomy and Soil Science*, 67, 1449-1464. <https://doi.org/10.1080/03650340.2020.1798000>
- Mrdakovic Popic, J., Meland, S., Salbu, B., & Skipperud, L. (2014). Mobility of Radionuclides and Trace Elements in Soil from Legacy NORM and Undisturbed Naturally 232-Th-Rich Sites. *Environmental Science: Processes & Impacts*, 16, 1124-1134. <https://doi.org/10.1039/c3em00569k>
- Nabelkova, J. (2012). Trace Metals in the Bed Sediment of Small Urban Streams. *The Open Environmental & Biological Monitoring Journal*, 5, 48-55. <https://doi.org/10.2174/1875040001205010048>
- Noulas, C., Tziouvalekas, M., & Karyotis, T. (2018). Zinc in Soils, Water and Food Crops. *The Journal of Trace Elements in Medicine and Biology*, 49, 252-260. <https://doi.org/10.1016/j.jtemb.2018.02.009>

- Oladipupo Azeez, J., Olabisi Hassan, A., & Blessing Olowoboko, T. (2018). Differential Sorption Behavior of Cadmium, Lead, Zinc, and Copper in Some Tropical Soils and Their Environmental Implications. *Communications in Soil Science and Plant Analysis*, 49, 1707-1718. <https://doi.org/10.1080/00103624.2018.1474908>
- Pahlevan-Sharif, S., Mura, P., & Wijesinghe, S. N. R. (2019). A Systematic Review of Systematic Reviews in Tourism. *Journal of Hospitality and Tourism Management*, 39, 158-165. <https://doi.org/10.1016/j.jhtm.2019.04.001>
- Pathak, P., Singh, D. N., Pandit, G. G., & Rakesh, R. R. (2014). Determination of Distribution Coefficient: A Critical Review. *International Journal of Environment and Waste Management*, 14, 27-64. <https://doi.org/10.1504/IJEWEM.2014.062980>
- Piri, M., Sepehr, E., & Rengel, Z. (2019). Citric acid Decreased and Humic Acid Increased Zn Sorption in Soils. *Geoderma*, 341, 39-45. <https://doi.org/10.1016/j.geoderma.2018.12.027>
- Rahman, H. H., Niemann, D., & Singh, D. (2020). Arsenic Exposure and Association with Hepatitis E IgG Antibodies. *Occupational Diseases and Environmental Medicine*, 8, 111-122. <https://doi.org/10.4236/odem.2020.83009>
- Ren, B., Wu, Y., Deng, D., Tang, X., & Li, H. (2020). Effect of Multiple Factors on the Adsorption of Cd in an Alluvial Soil from Xiba, China. *Journal of Contaminant Hydrology*, 232, Article ID: 103605. <https://doi.org/10.1016/j.jconhyd.2020.103605>
- Rezaei, M. J., Farahbakhsh, M., Shahbazi, K., & Marzi, M. (2021). Study of Cadmium Distribution Coefficient in Acidic and Calcareous Soils of Iran: Comparison between Low and High Concentrations. *Environmental Technology & Innovation*, 22, Article ID: 101516. <https://doi.org/10.1016/j.eti.2021.101516>
- Roth, E., Mancier, V., & Fabre, B. (2012). Adsorption of Cadmium on Different Granulometric Soil Fractions: Influence of Organic Matter and Temperature. *Geoderma*, 189-190, 133-143. <https://doi.org/10.1016/j.geoderma.2012.04.010>
- Sedeño-Díaz, J. E., López-López, E., Mendoza-Martínez, E., Rodríguez-Romero, A. J., & Morales-García, S. S. (2019). Distribution Coefficient and Metal Pollution Index in Water and Sediments: Proposal of a New Index for Ecological Risk Assessment of Metals. *Water*, 12, Article 29. <https://doi.org/10.3390/w12010029>
- Shaheen, S. M., Antoniadis, V., Kwon, E. E., Biswas, J. K., Wang, H., Ok, Y. S., & Rinklebe, J. (2017). Biosolids Application Affects the Competitive Sorption and Lability of Cadmium, Copper, Nickel, Lead, and Zinc in Fluvial and Calcareous Soils. *Environmental Geochemistry and Health*, 39, 1365-1379. <https://doi.org/10.1007/s10653-017-9927-4>
- Shaheen, S. M., Derbalah, A. S., & Moghanm, F. S. (2012). Removal of Heavy Metals from Aqueous Solution by Zeolite in Competitive Sorption System. *International Journal of Environmental Science and Development*, 3, 362-367. <https://doi.org/10.7763/IJESD.2012.V3.248>
- Shaheen, S. M., Tsadilas, C. D., & Rinklebe, J. (2013). A Review of the Distribution Coefficients of Trace Elements in Soils: Influence of Sorption System, Element Characteristics, and Soil Colloidal Properties. *Advances in Colloid and Interface Science*, 201-202, 43-56. <https://doi.org/10.1016/j.cis.2013.10.005>
- Shaheen, S. M., Tsadilas, C. D., Niazi, N. K., Hseu, Z.-Y., Ok, Y. S., Selim, M., & Rinklebe, J. (2018). Impact of Biosolid Application Rates on Competitive Sorption and Distribution Coefficients of Cd, Cu, Ni, Pb, and Zn in an Alfisol and an Entisol. *Process Safety and Environmental Protection*, 115, 38-48. <https://doi.org/10.1016/j.psep.2017.10.013>
- Shaheen, S. M., Tsadilas, C. D., Rupp, H., Rinklebe, J., & Meissner, R. (2015). Distribution Coefficients of Cadmium and Zinc in Different Soils in Mono-Metal and Competitive

- Sorption Systems. *Journal of Plant Nutrition and Soil Science*, 178, 671-681.
<https://doi.org/10.1002/jpln.201400475>
- Sharma, B. D., & Sharma, S. (2013). Effects of Organic Matter and Ionic Strength of Supporting Electrolyte on Zinc Adsorption in Benchmark Soils of Punjab in Northwest India. *Communications in Soil Science and Plant Analysis*, 44, 922-938.
<https://doi.org/10.1080/00103624.2012.747602>
- Sharma, B., Sarkar, A., Singh, P., & Singh, R. P. (2017). Agricultural Utilization of Biosolids: A Review on Potential Effects on Soil and Plant Growth. *Waste Management*, 64, 117-132. <https://doi.org/10.1016/j.wasman.2017.03.002>
- Soares, M. R., de Souza Sarkis, J. E., & Alleoni, L. R. F. (2021). Proposal of New Distribution Coefficients (Kd) of Potentially Toxic Elements in Soils for Improving Environmental Risk Assessment in the State of Sao Paulo, Southeastern Brazil. *Journal of Environmental Management*, 285, Article ID: 112044.
<https://doi.org/10.1016/j.jenvman.2021.112044>
- Sparks, D. L. (2003). *Environmental Soil Chemistry*. Academic Press.
<https://doi.org/10.1016/B978-012656446-4/50001-3>
- Strawn, D. G. (2018). Review of Interactions between Phosphorus and Arsenic in Soils from Four Case Studies. *Geochemical Transactions*, 19, Article No. 10.
<https://doi.org/10.1186/s12932-018-0055-6>
- Swati, A., & Hait, S. (2017). Fate and Bioavailability of Heavy Metals during Vermicomposting of Various Organic Wastes—A Review. *Process Safety and Environmental Protection*, 109, 30-45. <https://doi.org/10.1016/j.psep.2017.03.031>
- Tahervand, S., & Jalali, M. (2017). Sorption and Desorption of Potentially Toxic Metals (Cd, Cu, Ni and Zn) by Soil Amended with Bentonite, Calcite and Zeolite as a Function of pH. *Journal of Geochemical Exploration*, 181, 148-159.
<https://doi.org/10.1016/j.gexplo.2017.07.005>
- Ugochukwu, N., Mohamed, I., Ali, M., Iqbal, J., Fu, Q., Zhu, J., Jiang, G., & Hu, H. (2013). Impacts of Inorganic Ions and Temperature on Lead Adsorption onto Variable Charge Soils. *Catena*, 109, 103-109. <https://doi.org/10.1016/j.catena.2013.04.009>
- Urbaniak, M., Wyrwicka, A., Toloczko, W., Serwecinska, L., & Zielinski, M. (2017). The Effect of Sewage Sludge Application on Soil Properties and Willow (*Salix* sp.) Cultivation. *Science of the Total Environment*, 586, 66-75.
<https://doi.org/10.1016/j.scitotenv.2017.02.012>
- Vatandoost, M., Naghipour, D., Omid, S., & Ashrafi, S. D. (2018). Survey and Mapping of Heavy Metals in Groundwater Resources around the Region of the Anzali International Wetland; a Dataset. *Data in Brief*, 18, 463-469.
<https://doi.org/10.1016/j.dib.2018.03.058>
- Venegas, A., Rigol, A., & Vidal, M. (2015). Viability of Organic Wastes and Biochars as Amendments for the Remediation of Heavy Metal-Contaminated Soils. *Chemosphere*, 119, 190-198. <https://doi.org/10.1016/j.chemosphere.2014.06.009>
- Verbeeck, M., Thiry, Y., & Smolders, E. (2020). Soil Organic Matter Affects Arsenic and Antimony Sorption in Anaerobic Soils. *Environmental Pollution*, 257, Article ID: 113566.
<https://doi.org/10.1016/j.envpol.2019.113566>
- Vithanage, M., Herath, I., Joseph, S., Bundschuh, J., Bolan, N., Ok, Y. S., Kirkham, M. B., & Rinklebe, J. (2017). Interaction of Arsenic with Biochar in Soil and Water: A Critical Review. *Carbon*, 113, 219-230. <https://doi.org/10.1016/j.carbon.2016.11.032>
- Wang, F., Bao, K., Huang, C., Zhao, X., Han, W., & Yin, Z. (2022). Adsorption and pH Values Determine the Distribution of Cadmium in Terrestrial and Marine Soils in the Nansha Area, Pearl River Delta. *International Journal of Environmental Research and*

- Public Health*, 19, Article 793. <https://doi.org/10.3390/ijerph19020793>
- Wang, J., Lü, C., He, J., & Zhao, B. (2016). Binding Characteristics of Pb²⁺ to Natural Fulvic Acid Extracted from the Sediments in Lake Wuliangsu Hai, Inner Mongolia Plateau, P.R. China. *Environmental Earth Sciences*, 75, Article No. 768. <https://doi.org/10.1007/s12665-016-5608-3>
- Wang, J., Zeng, X., Zhang, H., Li, Y., Zhao, S., Bai, L., Su, S., & Wang, Y. (2018). Kinetic Release of Arsenic after Exogenous Inputs into Two Different Types of Soil. *Environmental Science and Pollution Research*, 25, 12876-12882. <https://doi.org/10.1007/s11356-018-1550-0>
- Wokhe, T. (2015). Heavy Metals Pollution of Water and Sediment in Mada River, Nigeria. *Journal of Scientific Research and Reports*, 6, 157-164. <https://doi.org/10.9734/JSRR/2015/14803>
- Yang, H., Huang, K., Zhang, K., Weng, Q., Zhang, H., & Wang, F. (2021). Predicting Heavy Metal Adsorption on Soil with Machine Learning and Mapping Global Distribution of Soil Adsorption Capacities. *Environmental Science & Technology*, 55, 14316-14328. <https://doi.org/10.1021/acs.est.1c02479>
- Yang, Y., Wang, M., Chen, W., Li, Y., & Peng, C. (2017). Cadmium Accumulation Risk in Vegetables and Rice in Southern China: Insights from Solid-Solution Partitioning and Plant Uptake Factor. *Journal of Agricultural and Food Chemistry*, 65, 5463-5469. <https://doi.org/10.1021/acs.jafc.7b01931>
- Yavar Ashayeri, N., & Keshavarzi, B. (2019). Geochemical Characteristics, Partitioning, Quantitative Source Apportionment, and Ecological and Health Risk of Heavy Metals in Sediments and Water: A Case Study in Shadegan Wetland, Iran. *Marine Pollution Bulletin*, 149, Article ID: 110495. <https://doi.org/10.1016/j.marpolbul.2019.110495>
- Yuan, H., Yin, H., Yang, Z., Yu, J., Liu, E., Li, Q., Tai, Z., & Cai, Y. (2020). Diffusion Kinetic Process of Heavy Metals in Lacustrine Sediment Assessed Under Different Redox Conditions by DGT and DIFS Model. *Science of the Total Environment*, 741, Article ID: 140418. <https://doi.org/10.1016/j.scitotenv.2020.140418>
- Yuan, Y., Xiang, M., Liu, C., & Theng, B. K. G. (2017). Geochemical Characteristics of Heavy Metal Contamination Induced by a Sudden Wastewater Discharge from a Smelter. *Journal of Geochemical Exploration*, 176, 33-41. <https://doi.org/10.1016/j.gexplo.2016.07.005>
- Zafeiriou, I., Gasparatos, D., Kalyvas, G., Ioannou, D., & Massas, I. (2019). Desorption of Arsenic from Calcareous Mine Affected Soils by Phosphate Fertilizers Application in Relation to Soil Properties and As Partitioning. *Soil Systems*, 3, Article 54. <https://doi.org/10.3390/soilsystems3030054>
- Zakari, S., Jiang, X., Zhu, X., Liu, W., Allakonon, M. G. B., Singh, A. K., Chen, C., Zou, X., Akponikpe, P. B. I., Dossa, G. G. O., & Yang, B. (2021). Influence of Sulfur Amendments on Heavy Metals Phytoextraction from Agricultural Contaminated Soils: A Meta-Analysis. *Environmental Pollution*, 288, Article ID: 117820. <https://doi.org/10.1016/j.envpol.2021.117820>
- Zhang, J., Zhou, F., Chen, C., Sun, X., Shi, Y., Zhao, H., & Chen, F. (2018). Spatial Distribution and Correlation Characteristics of Heavy Metals in the Seawater, Suspended Particulate Matter and Sediments in Zhanjiang Bay, China. *PLOS ONE*, 13, e0201414. <https://doi.org/10.1371/journal.pone.0201414>
- Zhang, W., Rao, W., Li, L., Brusseau, M. L., & Liu, Y. (2021). Adsorption of Cadmium onto Sand-Attapulgitic Cutoff Wall Backfill Media. *Water, Air, & Soil Pollution*, 232, Article No. 47. <https://doi.org/10.1007/s11270-021-04981-z>
- Zhao, X., Jiang, T., & Du, B. (2014). Effect of Organic Matter and Calcium Carbonate on

Behaviors of Cadmium Adsorption-Desorption on/from Purple Paddy Soils. *Chemosphere*, 99, 41-48. <https://doi.org/10.1016/j.chemosphere.2013.09.030>

Zheng, J. C., Liu, H. Q., Feng, H. M., Li, W. W., Lam, M. H., Lam, P. K., & Yu, H. Q. (2016). Competitive Sorption of Heavy Metals by Water Hyacinth Roots. *Environmental Pollution*, 219, 837-845. <https://doi.org/10.1016/j.envpol.2016.08.001>

Zhuang, W., & Gao, X. (2013). Acid-Volatile Sulfide and Simultaneously Extracted Metals in Surface Sediments of the Southwestern Coastal Laizhou Bay, Bohai Sea: Concentrations, Spatial Distributions and the Indication of Heavy Metal Pollution Status. *Marine Pollution Bulletin*, 76, 128-138. <https://doi.org/10.1016/j.marpolbul.2013.09.016>