

Article

Comparison of Arsenic Adsorption and Desorption Performance by Different Microaggregates in Black Soil

Lijuan Huo ^{1,2}, Peipei Zhang ^{1,2}, Jiahao Liu ^{1,2}, Rui Yang ^{1,2}, Qian Zhang ³, Shuting Tian ^{1,2}, Ting Wang ^{1,2}, Gaiqiang Yang ^{1,2,*}  and Hongqin Guo ^{2,*}

¹ School of Environment and Resources, Taiyuan University of Science and Technology, Taiyuan 030024, China; hlj@tyust.edu.cn (L.H.)

² Engineering Technology Innovation Center for Ecological Protection and Restoration in the Middle Yellow River, Ministry of Natural Resources, Taiyuan 030024, China

³ Standard Technology Engineering (Qing Dao) Co., Ltd., Qingdao 266000, China

* Correspondence: ygq@tyust.edu.cn (G.Y.); guohq2026@163.com (H.G.)

Abstract

The adsorption and desorption behavior of arsenic (As) in agricultural soils is a critical process controlling its migration, transformation, and bioavailability, with direct implications for food safety and environmental risk. Although black soil regions are major grain-producing areas in China, the roles of different soil microaggregate fractions and their components in As retention remain poorly understood. Therefore, batch equilibrium adsorption experiments were performed to study the adsorption and desorption behaviors of As(V) on the different microaggregates and to explore the effects of soil particle size, organic matter and iron oxide on the adsorption performance of As(V). The results show that as the concentration of As(V) increases, the adsorption capacity gradually reaches equilibrium, and the Freundlich equation fits the results well. The order of the As adsorption capacity of microaggregates of different particle sizes in black soil is as follows: (<0.002 mm) > 0.005 – 0.05 mm > 0.002 – 0.005 mm > 0.05 – 0.25 mm > 0.25 – 2 mm. The maximum adsorption capacity occurs in the microaggregates of the soil with the smallest particle size. The order of the As desorption capacity of microaggregates of each particle size is opposite to their adsorption capacity and inversely proportional to the content of each component in the soil. Removal of soil organic matter (SOM) and free iron oxide (Fe_d) significantly reduced the specific adsorption and immobilization capacity of black soil for As(V), while enhancing non-specific adsorption. This study elucidates the differential contributions of soil microaggregates and key components to As(V) retention and provides an experimental foundation for further research on the occurrence and migration mechanisms of As in black soil.

Keywords: soil; microaggregates; adsorption; desorption; As(V); soil components



Received: 17 January 2026

Revised: 20 February 2026

Accepted: 26 February 2026

Published: 28 February 2026

Copyright: © 2026 by the authors.

Licensee MDPI, Basel, Switzerland.

This article is an open access article distributed under the terms and conditions of the [Creative Commons Attribution \(CC BY\)](https://creativecommons.org/licenses/by/4.0/) license.

1. Introduction

Arsenic (As) is a highly toxic metalloid widely distributed in the environment. Its sources in soil include both natural processes—such as volcanic eruptions and pedogenesis—and anthropogenic activities, including mining and smelting of As-bearing ores and improper disposal of As-containing industrial wastewater [1]. As exists primarily in two oxidation states, As(III) and As(V), with As(III) being significantly more toxic. While As(III) predominates in aquatic systems, As(V) is the dominant species in aerobic soils [2–4]. As contamination in soil is persistent, prone to bioaccumulation, and often irreversible

due to strong binding with soil minerals. Its lack of visible indicators further complicates early detection and remediation, posing substantial long-term management challenges. [5]. Moreover, As can enter the human body through the food chain, leading to serious health risks [6].

The black soil region in Northeast China is a major grain-producing area [7]. With the rapid development of industry and agriculture, this region is also facing an increasingly severe risk of metallic element pollution [8,9]. Spatial heterogeneity in soil As content is pronounced, and similar contamination levels have been reported globally. For instance, Yang et al. [10] identified that over 50% of the Mississippi River Basin is at high risk of As pollution. In Bangladesh, irrigation with As-contaminated groundwater has led to severe soil As accumulation [11]. In Brazil, Feitosa et al. [12] reported soil As concentrations ranging from 240 to 380 mg·L⁻¹ in the Paracatu agricultural area, with significant variation across different farming zones.

After exogenous As enters the soil, it accumulates through adsorption, coprecipitation and complexation, leading to excessive soil As concentration and posing a great threat to human production and life. The adsorption and desorption processes of As on the soil surface are the main processes for controlling the migration of low-concentration As. Some studies have pointed out that the proportion of clay particles in the soil is positively correlated with the As content in the soil, and the As content in fine soil particles is higher than that in other particle sizes [13]. The specific surface area of soil increases as the particle size decreases, leading to an increase in the content of clay minerals and metal oxides per unit mass, which in turn enhances the specific adsorption capacity. This enhancement effect is not conducive to the desorption process of As(V) [14]. Tani et al. [15] demonstrated that manganese oxides can have a certain impact on the valence state of As. Manganese oxides act as electron acceptors during the oxidation process, oxidizing As(III) to As(V) or reducing the mobility of As in the soil by directly adsorbing As(V) [16]. The increase in the content of free iron oxide (Fe_d) in the soil can increase the number of adsorption sites in the soil, thereby enhancing the soil's adsorption capacity for As. When its content is high, the desorption amount is low [14]. The protonation and complexation reactions of functional groups such as carboxyl, phenolic hydroxyl, carbonyl and methoxy groups contained in soil organic matter (SOM) can affect the adsorption behavior of soil for As(V) [17]. Humic acid and fulvic acid, as active organic matter components in soil, have an inhibitory effect on the adsorption of As. However, as the adsorption time increases, these organic substances are gradually oxidized or decomposed by microorganisms, resulting in a weakened inhibitory effect and ultimately only changing the adsorption rate [18].

Other factors influencing As adsorption in soil include redox potential, pH, and microbial activity [19,20]. Soil aggregates, as fundamental structural units, bind sand, silt, and clay particles together with organic and inorganic substances [21–25]. The adsorption and bioavailability of As vary across aggregate sizes [26–28]. Smaller aggregates exhibit higher mobility and, once laden with heavy metals, pose greater pollution risks [29,30]. Although differences in As distribution and bioavailability among aggregate fractions have been reported in non-agricultural soils [31], systematic investigations focusing on black soil, a unique and agriculturally vital resource, remain scarce. In particular, the quantitative contributions and mechanistic roles of SOM and Fe_d in regulating As(V) adsorption–desorption across distinct microaggregate fractions have not been clearly established. Therefore, this study aims to: (1) isolate and characterize different microaggregate fractions from typical black soil; (2) quantify and compare their adsorption–desorption capacities and affinities for As(V); (3) elucidate the individual and combined effects of SOM and Fe_d on these processes through component removal experiments. The findings will provide a mechanistic understanding of As(V) retention at the microaggregate scale and offer

a theoretical basis for As pollution risk assessment and targeted remediation in black soil regions.

2. Materials and Methods

2.1. Chemicals and Equipment

All chemicals used were of analytical grade. Sodium arsenate heptahydrate ($\text{Na}_2\text{HAsO}_4 \cdot 7\text{H}_2\text{O}$; Sinopharm; purity $\geq 98\%$), sodium chloride (NaCl ; CAS 7647-14-5), hydrogen peroxide (H_2O_2 ; 30%), sodium citrate dihydrate ($\text{C}_6\text{H}_5\text{Na}_3\text{O}_7 \cdot 2\text{H}_2\text{O}$), sodium bicarbonate (NaHCO_3), and solid sodium hyposulfite ($\text{Na}_2\text{S}_2\text{O}_4$) were purchased from Sinopharm Chemical Reagent Co., Ltd. (Shanghai, China).

Equipment included a pH meter (S200, Mettler Toledo Instruments Shanghai Co., Ltd., Shanghai, China) and an ultrasonic cleaner (KQ-500DE, China Limited). Soil vibration and desorption experiments were conducted in a CNC oscillation incubator (HZQ-X100, Jin tan Jieruier Electric Appliance Co., Ltd., Changzhou, China). Centrifugation was carried out using a high-speed centrifuge (TG16WS, Cence Lab China Limited, Changsha, China). The concentration of As(V) in the solution was determined by atomic fluorescence spectroscopy (AFS-830 Beijing Jitian Instrument Co., Ltd., Beijing, China).

2.2. Test Soil Sample

The soil samples used in this study were collected from the surrounding farmlands of Bayan County, Heilongjiang Province. The parent material of the soil was dolomite weathering. The sampling depth was 0 to 20 cm. After air-drying and grinding, the samples were passed through a 2 mm nylon sieve for later use. Parts of them were taken to remove SOM and Fe_d , respectively, to obtain the soil microaggregates after removing the components, which were air-dried and ground for later use. The specific methods are as follows: For SOM removal, place the soil sample in a beaker, moisten it with deionized water, then add a specific mass fraction of H_2O_2 solution drop by drop and repeat the treatment 2 to 3 times until the color of the soil sample becomes lighter. Continue to add H_2O_2 until no bubbles emerge and the soil with SOM removed is obtained. Heat the obtained soil and wash the soil sample in the beaker with deionized water. The required samples can be obtained after drying [32]. For Fe_d removal, put a certain amount of the test soil into a conical flask, add $\text{C}_6\text{H}_5\text{Na}_3\text{O}_7 \cdot 2\text{H}_2\text{O}$ and NaHCO_3 of specific concentrations, then put it into a water bath pot and stir. Heat it to 90°C and keep it at that temperature for 2 h. Immediately add a specific mass of $\text{Na}_2\text{S}_2\text{O}_4$, stir for 15 min, then take it out, let it stand and pour out the supernatant. Repeat the above operation twice. The residue can be washed with $1\text{ mol}\cdot\text{L}^{-1}$ NaCl solution to obtain soil with Fe_d removed [33].

Another part of the soil sample was used to separate the soil components by particle size, following established extraction methods [34]. First, the soil sample was passed through a 0.25 mm sieve. The soil sample was then thoroughly mixed with deionized water, soaked overnight, and subjected to ultrasonic dispersion. The wet sieve method was used to separate particles of different sizes, which were (<0.002 mm), 0.002–0.005 mm, 0.005–0.05 mm, 0.05–0.25 mm and 0.25–2 mm. The extracted soil microaggregates of each particle size were collected for use. Meanwhile, the physicochemical properties of the soil microaggregates were determined [32].

2.3. Experimental Methods

2.3.1. Adsorption–Desorption Experiments of Different Microaggregates with As(V)

Exactly 1.00 g of each microaggregate sample was weighed into 50 mL centrifuge tubes, and 20 mL of As(V) solution (prepared from $\text{Na}_2\text{HAsO}_4 \cdot 7\text{H}_2\text{O}$) at concentrations of 0, 5, 10, 20, 40, 80, 120, 140, and 160 $\text{mg}\cdot\text{L}^{-1}$ was added, with $0.01\text{ mol}\cdot\text{L}^{-1}$ NaCl

as the background electrolyte. The tubes were shaken at 180 rpm and 25 °C for 48 h. After centrifugation, the supernatant was collected for As(V) analysis. All treatments were performed in triplicate. The pH of the reaction system ranged from 6.3 to 6.5, and no reducing agents were added. Experiments were conducted under ambient pressure and in the dark to minimize As(V) reduction. Samples were analyzed immediately after centrifugation to prevent speciation changes.

Immediately following adsorption, the supernatant was removed, and 20 mL of 0.01 mol·L⁻¹ NaCl solution was added to the residue. The tubes were shaken at 180 rpm for 48 h. This process was repeated three times to reach desorption equilibrium. The final supernatant was collected, filtered, and analyzed for As(V) concentration.

The adsorption capacity q_e (mg·g⁻¹) was calculated as

$$q_e = \frac{(C_0 - C_e)V}{m}. \quad (1)$$

In the formula,

V : Total volume of the mixed solution (L);

C_0 : Adsorption of As concentration in the initial solution (mg·L⁻¹);

C_e : Solution concentration at adsorption equilibrium (mg·L⁻¹);

m : Soil sample mass (g).

The desorption capacity q_d (mg·g⁻¹) of As(V) can be calculated according to the following formula:

$$q_d = \frac{C_{de}(20 + W_2 - W_1) - C_e(W_2 - W_1)}{m}. \quad (2)$$

$$\text{Desorption rate (\%)} = \frac{q_d}{q_e} \times 100\%. \quad (3)$$

In the formula,

C_{de} : The concentration of As in the desorption equilibrium liquid (mg·L⁻¹);

C_e : The concentration of As in the adsorption equilibrium solution (mg·L⁻¹);

W_1 : The total weight of the centrifuge tube and soil sample (g);

W_2 : The total weight of the centrifuge tube, soil sample and residual liquid (g);

20: The volume of liquid added in the experiment (mL);

m : The soil sample mass (g).

2.3.2. Data Processing

To calculate the relevant parameters of As(V) adsorption in soil, the Langmuir and Freundlich equations were used to fit the experimental data. The following are the fitting equations:

Langmuir equation:

$$q_e = \frac{Q_m \cdot b \cdot C_e}{1 + b \cdot C_e}. \quad (4)$$

Freundlich equation:

$$Q_e = k \cdot C_e^{(\frac{1}{n})} \quad (5)$$

Among them,

Q_m : The theoretical maximum adsorption capacity (mg·g⁻¹);

C_e : The mass concentration of the equilibrium liquid (mg·L⁻¹);

k , n , and b : Constants related to adsorption equilibrium.

Pearson correlation analysis was performed to examine the relationships between soil physicochemical properties and adsorption parameters. Correlation coefficients (r) were calculated, and the significance of the correlations was assessed at the $p < 0.05$ level.

The distribution coefficient (F ; $\text{L} \cdot \text{g}^{-1}$), representing the affinity of the soil for As(V) at equilibrium, was calculated as follows:

$$F = \frac{q_e}{C_e} \quad (6)$$

where q_e is the adsorption capacity at equilibrium ($\text{mg} \cdot \text{g}^{-1}$) and C_e is the equilibrium concentration of As(V) in the solution ($\text{mg} \cdot \text{L}^{-1}$). A higher F value indicates a stronger partitioning of As(V) to the solid phase, reflecting a greater overall adsorptive retention capacity [35].

3. Results and Analysis

3.1. Physicochemical Properties of Bulk Soil and Microaggregates

The mass distribution and physicochemical properties of different microaggregate fractions are presented in Table 1. The 0.05–0.25 mm, 0.005–0.05 mm, and 0.002–0.005 mm fractions accounted for the larger proportions, while the 0.25–2 mm and (<0.002 mm) fractions were minor. The pH varied little among fractions. The contents of SOM, Fe_d and cation exchange capacity (CEC) followed a consistent order: (<0.002 mm) > 0.005–0.05 mm > 0.002–0.005 mm > bulk soil > 0.25–2 mm > 0.05–0.25 mm.

Table 1. Basic physicochemical properties of bulk soil and microaggregate fractions.

Granulometric Class (mm)	pH	SOM ($\text{g} \cdot \text{kg}^{-1}$)	CEC ($\text{cmol} \cdot \text{kg}^{-1}$)	Fe_d ($\text{g} \cdot \text{kg}^{-1}$)	Mass Fraction (%)
original soil	6.51 ± 0.02^a	21.04 ± 0.51^c	16.17 ± 0.43^d	17.42 ± 0.39^c	—
0.25–2	6.52 ± 0.03^a	20.25 ± 0.48^d	15.99 ± 0.41^e	16.08 ± 0.36^d	5.03 ± 0.12^c
0.05–0.25	6.47 ± 0.02^b	18.65 ± 0.45^e	14.95 ± 0.38^f	15.44 ± 0.34^e	38.17 ± 0.23^a
0.005–0.05	6.39 ± 0.03^c	31.16 ± 0.37^b	19.07 ± 0.47^b	20.16 ± 0.41^b	36.94 ± 0.31^a
0.002–0.005	6.31 ± 0.02^d	21.11 ± 0.42^c	17.51 ± 0.45^c	17.6 ± 0.42^c	13.89 ± 0.31^b
<0.002	6.37 ± 0.03^c	75.25 ± 0.82^a	29.38 ± 0.68^a	35.92 ± 0.63^a	5.97 ± 0.14^c

Note: Different superscript lowercase letters in the same column indicate significant differences among different particle size fractions ($p < 0.05$). The mass fraction column applies only to the particle size fractions, with the corresponding entry for bulk soil denoted as “—”.

3.2. Adsorption and Desorption Behavior of As(V) on Different Microaggregates

3.2.1. The Adsorption Behavior of Microaggregates of Different Particle Sizes for As(V)

The adsorption capacity of all microaggregate fractions increased with increasing equilibrium As(V) concentration, but the patterns varied by particle size. At $80 \text{ mg} \cdot \text{L}^{-1}$, adsorption by the 0.05–0.25 mm and 0.25–2 mm fractions approached saturation. For the 0.005–0.05 mm and 0.002–0.005 mm fractions, adsorption continued to increase beyond $80 \text{ mg} \cdot \text{L}^{-1}$, albeit at a reduced rate. For the (<0.002 mm) fraction, the adsorption rate began to slow at approximately $40 \text{ mg} \cdot \text{L}^{-1}$ ($p < 0.05$), suggesting saturation of high-affinity sites followed by adsorption onto lower-affinity sites. The order of adsorption capacity of As(V) by microaggregates of different particle sizes of black soil is as follows: (<0.002 mm) > 0.005–0.05 mm > 0.002–0.005 mm > 0.05–0.25 mm > 0.25–2 mm. The maximum adsorption capacity of black soil of different particle sizes ranges from 1.14 to $2.21 \text{ mg} \cdot \text{g}^{-1}$ (Figure 1). There are significant differences in the variation trends of the adsorption capacity of As(V) by black soil of different particle sizes. The adsorption effect is the best for (<0.002 mm), and the adsorption effects of 0.002–0.005 mm and 0.005–0.05 mm are relatively similar, while the adsorption effects of 0.05–0.25 mm and 0.25–2 mm are relatively poor.

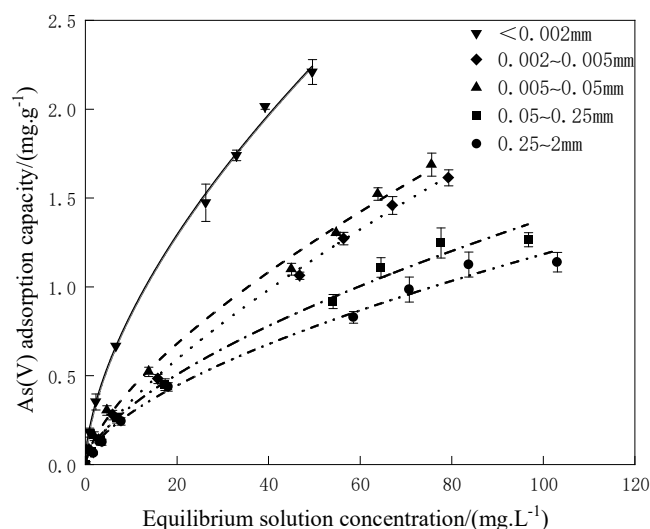


Figure 1. Isothermal adsorption curves of As(V) on black soil microaggregates of different particle sizes (25 °C; 0.01 mol·L^{−1} NaCl).

The Langmuir and Freundlich equations were used to fit the above process (Table 1). The results show that both equations can well describe the adsorption behavior of As(V) on the microaggregates of black soil at various particle sizes, with r^2 reaching above 0.9871. Among them, the fitting effect of the Freundlich equation is the best. The magnitude relationship between the maximum adsorption capacity of the original soil and microaggregates of different particle sizes for As(V), which can be obtained through the Langmuir equation, is as follows: 0.002–0.005 mm > (<0.002 mm) $\approx$ (0.005–0.05 mm) > 0.05–0.25 mm > 0.25–2 mm. In the Freundlich equation, n and k are constants. k is an empirical constant commonly used to represent the relative adsorption capacity or affinity in heterogeneous systems such as soil [36]. Under research conditions, k can be used to describe the adsorption capacity of soil for As, with higher k values typically corresponding to larger adsorption capacities. The soil with a particle size of (<0.002 mm) exhibited the highest k value. In contrast, the soil within the particle size range of 0.002–0.005 mm showed the lowest k value, indicating that the adsorption capacity of the tested component was relatively low.

Further analysis of the correlation between the adsorption k values of microaggregates of different particle sizes and the physicochemical properties of the soil revealed that the k values were positively correlated with the contents of SOM, Fe_d, and CEC, with correlation coefficients of 0.9842, 0.9710, and 0.9421 respectively (Figure 2a). With the increase in the content of SOM, Fe_d, and CEC in the soil, the adsorption capacity of black soil for As(V) also increased. Figure 2b illustrates that the maximum adsorption capacity of microaggregates of different particle sizes for As(V) is positively correlated with the contents of SOM, Fe_d, and CEC, indicating that regarding the content of heavy metals, the higher the content, the greater the adsorption capacity.

The distribution coefficient F value reflected by soil adsorption of As(V), that is, the ratio of the adsorption amount of As(V) by microaggregates of different particle sizes at adsorption equilibrium to the concentration of As(V) in the equilibrium solution, can also reflect the adsorption and fixation capacity of soil for As(V) [37]. Figure 3 demonstrates that the soil with a particle size of (<0.002 mm) has the highest adsorption capacity for As(V) and the strongest ability to adsorb and immobilize it.

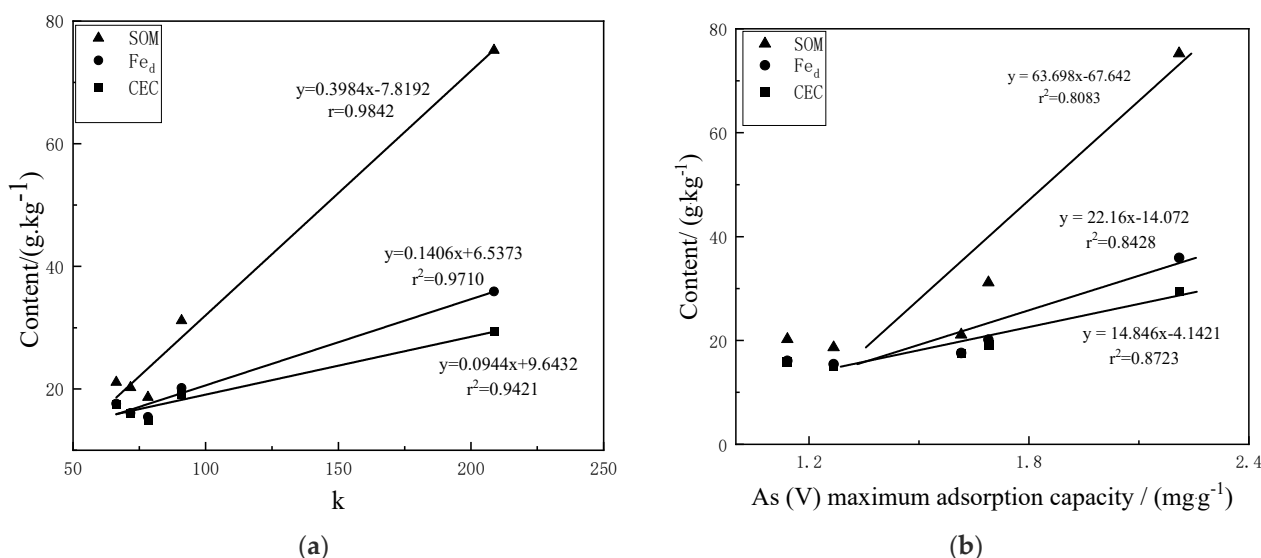


Figure 2. Correlations between (a) Freundlich constant k and (b) maximum adsorption capacity Q_m with soil properties for different microaggregate fractions.

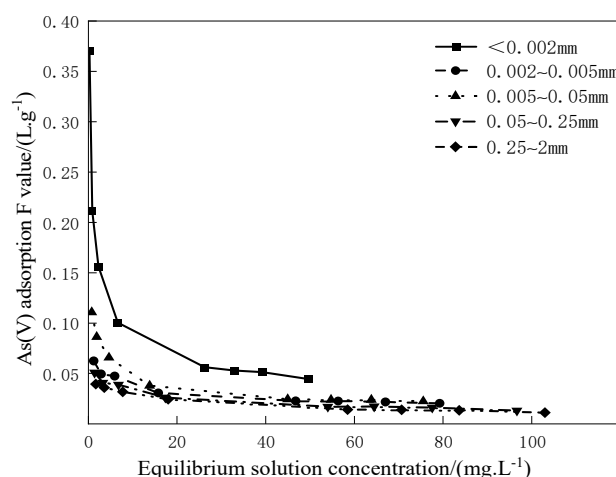


Figure 3. Relationship between distribution coefficient (F) of As(V) and equilibrium solution concentration for microaggregates of different particle sizes.

3.2.2. Analysis of Desorption Characteristics of Microaggregates of Different Particle Sizes

Studies have shown that non-specifically adsorbed As(V) can be partially or completely desorbed by neutral salt solutions, while specifically adsorbed As(V) ions are not easily desorbed [38]. The desorption capacity (rate) can be used as an indicator to judge the degree of desorption, i.e., to measure the soil's ability to hold ions. The desorption characteristics of microaggregates of different particle sizes for As(V) can be described by the easy desorption capacity (rate) and the difficult desorption capacity (rate). The low desorption rates, particularly for the (< 0.002 mm) fraction, underscore that As(V) retention is dominated by specific adsorption (chemisorption). This primarily involves the formation of inner-sphere complexes with iron oxide surfaces (e.g., ferrihydrite and goethite) present in Fe_d. As(V) oxyanions ($H_2AsO_4^-$ or $HAsO_4^{2-}$) undergo ligand exchange with surface hydroxyl groups, forming monodentate or bidentate complexes, which are highly stable and not easily desorbed by indifferent electrolytes [39,40].

The desorption process of As(V) by microaggregates of different particle sizes in black soil showed that after shaking for 48 h, the desorption basically reached equilibrium (Figure 4). The order of the desorption rates of soil of each grain size is $0.25\text{--}2\text{ mm} > 0.05\text{--}0.25\text{ mm} > 0.002\text{--}0.005\text{ mm} > 0.005\text{--}0.05\text{ mm} > (< 0.002\text{ mm})$, among

which the desorption rate of 0.25–2 mm is the highest, reaching 33.89%. The desorption rate of (<0.002 mm) microaggregates was the lowest at only 6.24%. The order of the difficult desorption rate is opposite to that of the easy desorption rate (Figure 4b). The research results show that the adsorption of As(V) by black soil is mainly specific adsorption, supplemented by non-specific adsorption. Moreover, as the concentration of exogenous As(V) increases, the adsorption capacity of both adsorption methods shows an increasing trend. Figure 4a reveals that the changes in the desorption amount of As(V) by soils of different particle sizes show differences, among which the relationship between the size of the desorption amount and the desorption rate remains consistent. The soil with a particle size of 0.25–2 mm showed the most obvious trend of desorption amount variation, with the maximum desorption amount being $0.3866 \text{ mg}\cdot\text{g}^{-1}$. The desorption capacity variation trend of the particle size (<0.002 mm) is relatively gentle, and the maximum desorption capacity is $0.1379 \text{ mg}\cdot\text{g}^{-1}$. The desorption capacity of other grain sizes of soil lies between these two (Figure 4a).

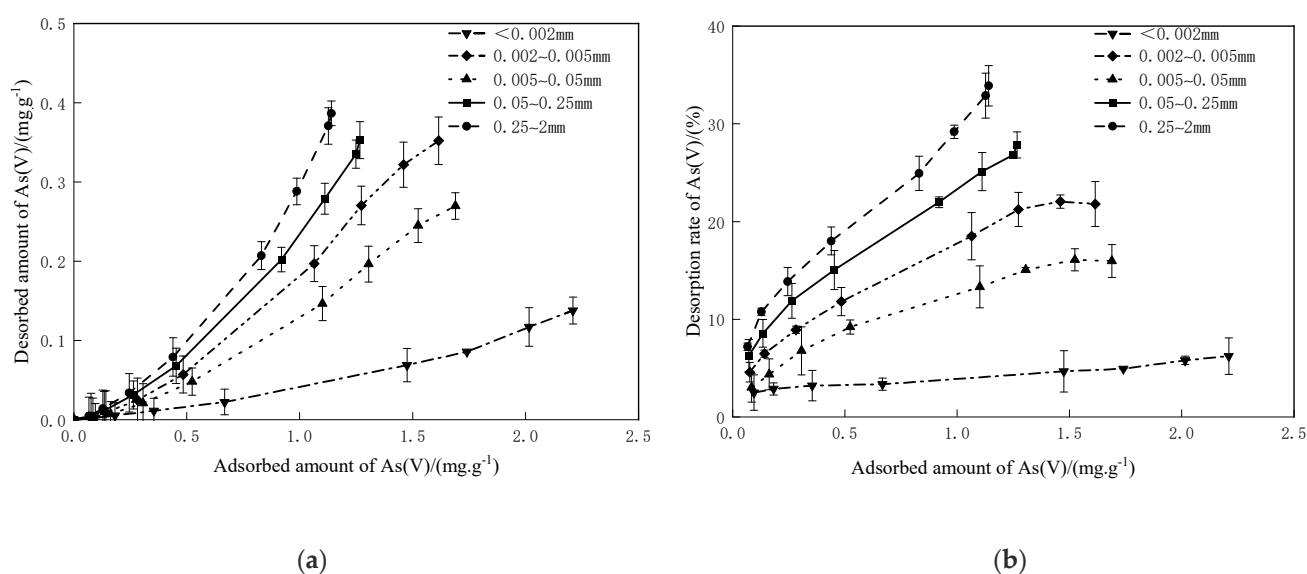


Figure 4. (a) Desorption isotherms and (b) desorption rates of As(V) on different microaggregate fractions.

3.3. Analysis of the Adsorption–Desorption Characteristics of the Original Soil for As(V) After the Removal of Soil Components

3.3.1. Analysis of Original Soil Adsorption Characteristics of As(V) After Removal of Soil Components

Figure 5 shows the changes in the adsorption capacity of As(V) by black soil before and after treatment (including the removal of SOM and Fe_d). With the increase in the equilibrium liquid concentration, the adsorption capacity of As(V) by black soil also shows an upward trend, which is significantly different from the changes in adsorption capacity before the removal of soil components. Compared with the untreated soil, the difference in the changing trend of soil adsorption capacity after the removal of SOM and Fe_d was reduced, and the adsorption capacity decreased significantly. Before the removal of components, the adsorption capacity of As(V) was almost in equilibrium after the concentration of the equilibrium solution reached $80 \text{ mg}\cdot\text{L}^{-1}$. After the removal of SOM and Fe_d , the adsorption equilibrium concentration of soil for As(V) tended to equilibrium only after $120 \text{ mg}\cdot\text{L}^{-1}$. Therefore, after the components of the soil are removed, the equilibrium concentration required for it to reach the maximum adsorption capacity lags behind.

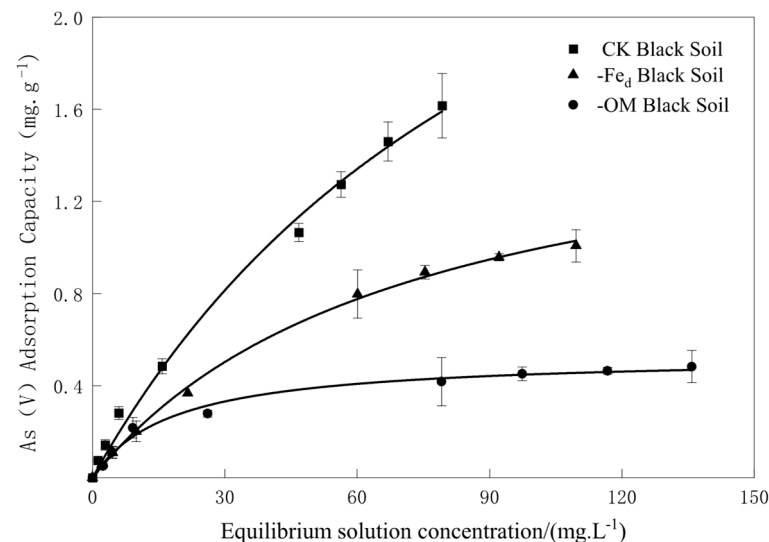


Figure 5. Isothermal adsorption curves of As(V) by the original soil after the removal of soil components.

The Langmuir and Freundlich equations were used to fit the adsorption process. The specific fitting parameters are shown in Table 2. Under different processing methods, both equations can fit well, with R^2 reaching above 0.9659. Among them, the fitting effect of the Langmuir equation is the best. The Q_m value after removing Fe_d is greater than that after removing SOM, indicating that organic matter has a greater impact on the saturated adsorption capacity of the soil. According to the adsorption constant k and n values obtained from the Freundlich equation, it can be known that the k value after removing Fe_d is smaller than that of the unremoved components, indicating that the removal of Fe_d has a greater impact on the adsorption of heavy metals in the soil than the removal of SOM. In conclusion, by removing the components, the adsorption capacity of black soil for As(V) decreased sharply, indicating that SOM and Fe_d are important indicators affecting the adsorption capacity of soil [14].

Table 2. Fitting parameters of the isothermal adsorption equation for As(V) of microaggregates of different particle sizes and the original soil after the removal of soil components.

Equation		Langmuir		Freundlich		
Parameter	b	Q_m (mg.g ⁻¹)	r^2	k	n	r^2
<0.002 mm	0.0317	3.8005	0.9911	0.2087	1.6443	0.9986
0.002~0.005 mm	0.0091	3.5265	0.9949	0.0663	1.3679	0.9988
0.005~0.05 mm	0.0121	3.4079	0.9871	0.0909	1.4889	0.9966
0.05~0.25 mm	0.0168	2.0888	0.9931	0.0782	1.6040	0.9919
0.25~2 mm	0.0180	1.7689	0.9934	0.0717	1.6423	0.9932
Before Removal	0.0091	3.8005	0.9949	0.0663	1.3679	0.9988
SOM Removal	0.0966	0.7431	0.9989	0.1427	2.9423	0.9659
Fe_d Removal	0.0140	1.6999	0.9985	0.0524	1.5556	0.9901

3.3.2. Analysis of the Desorption Characteristics of As(V) in the Original Soil After the Removal of Soil Components

After the removal of SOM and Fe_d , the easily desorbed amount of As(V) by black soil showed a gradually increasing trend, which was higher than that before the removal of components (Figure 6). The results show that after the removal of components, the overall adsorption and fixation capacity of black soil for As(V) decreases, and at the same time, its buffering capacity for As(V) also declines. In terms of adsorption mode, this change leads

to a relatively increased non-specific adsorption capacity of the soil for As(V), while the specific adsorption capacity is relatively weakened [40]. The amount of easily desorbed substances showed an upward trend to a certain degree, with the removal of SOM showing the largest trend, followed by the removal of Fe_d . Moreover, SOM had a greater impact on the specific adsorption capacity of soil than Fe_d . Figure 6b shows that the maximum difficult desorption rate of black soil changed significantly before and after the removal of soil components. After the removal of SOM, the ratio of the desorption rate of black soil increased from 13.85% to 31.13%. After the removal of Fe_d , the desorption rate of black soil increased from 6.73% to 19.52%. It can be seen that the influence of SOM on the specific adsorption of soil is slightly greater than that of Fe_d .

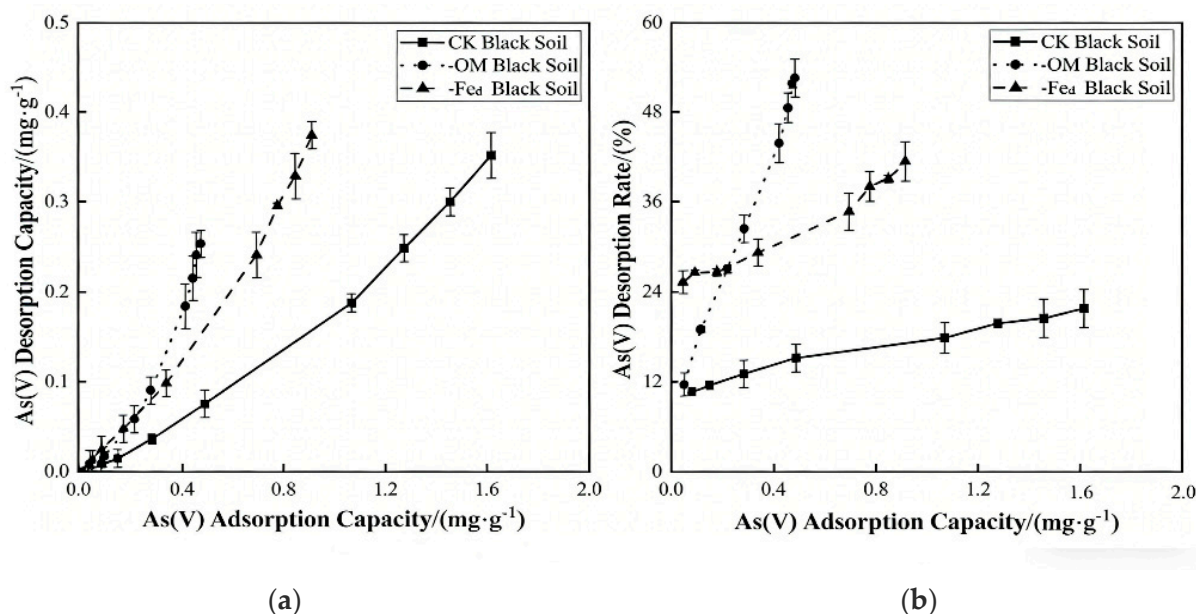


Figure 6. Isothermal desorption amounts (a) and desorption rates (b) of As(V) by the original soil after the removal of soil components.

4. Discussion

The research results show that with the increase in the equilibrium liquid concentration, the adsorption capacity of black soil microaggregates of different particle sizes for As(V) shows an increasing trend, which is manifested as $(<0.002 \text{ mm}) > 0.005\text{--}0.05 \text{ mm} > 0.002\text{--}0.005 \text{ mm} > 0.05\text{--}0.25 \text{ mm} > 0.25\text{--}2 \text{ mm}$. The microaggregates of the smallest particle size ($<0.002 \text{ mm}$) have the highest adsorption capacity (up to $2.21 \text{ mg}\cdot\text{g}^{-1}$), and the variation trends of adsorption capacity among microaggregates of different particle sizes show differences. This superior performance is attributed to the combined effects of large specific surface area and enrichment of reactive components such as SOM and Fe_d . The high specific surface area provides abundant contact sites for As(V) oxyanions [38,41], while SOM and Fe_d supply oxygen-containing functional groups (e.g., carboxyl and phenolic $-\text{OH}$) and surface hydroxyl groups that facilitate ligand exchange and inner-sphere complex formation [42]. The high CEC of the fine fraction also contributes to electrostatic (non-specific) adsorption. These physicochemical characteristics make the ($<0.002 \text{ mm}$) microaggregates the most effective sink for As(V) in black soil. This difference is mainly attributed to the different adsorption capacities of soil of each particle size, reflecting that the decrease in soil particle size is accompanied by an increase in specific surface area. This further enhances the adsorption viscosity and adsorption efficiency. However, within the slightly larger particle size range of $0.002\text{--}0.05 \text{ mm}$, the opposite adsorption behavior was observed. The adsorption capacity of soil with particle sizes

ranging from 0.005 to 0.05 mm is greater than that of soil with particle sizes ranging from 0.002 to 0.005 mm, indicating that the adsorption capacity may also be closely related to the specific surface area, the density of reactive sites (e.g., on clay minerals and oxides), and the pore structure within and between aggregates. Soils with higher clay content typically possess a greater abundance of adsorbents like iron/aluminum oxides and layer silicate clays, which provide specific adsorption sites for As(V) oxyanions. Furthermore, a more intricate network of micro-pores in finer aggregates can enhance solute contact time and increase the diffusion path length, potentially promoting adsorption [42]. Clay soil has poor permeability and a strong adsorption capacity [43]. When the concentration of the equilibrium liquid is relatively low, the growth rate of the adsorption capacity of each particle size of microaggregate for As(V) is significantly higher than that at the high-concentration stage, showing an increasing pattern that is initially rapid and then gradually slows down. This phenomenon might be due to the fact that within the low concentration range, the surface of the microaggregates contains a large number of adsorption sites with strong affinity for As(V). However, as the concentration increases, these high-affinity adsorption sites gradually become saturated, and As(V) is adsorbed by low-affinity sites instead. At this point, the specific adsorption effect of the soil on As(V) gradually strengthens until it slowly reaches the adsorption equilibrium state. Generally speaking, if the contribution from non-specific adsorption (e.g., electrostatic attraction) predominates over that from specific adsorption (e.g., ligand exchange), the adsorption process may exhibit a characteristic pattern of rapid initial increase followed by gradual stabilization. The F values of microaggregates of different particle sizes are negatively correlated with the concentration of the equilibrium liquid. This phenomenon is mainly attributed to the fact that as the adsorption capacity of soil microaggregates of each particle size for As(V) increases, their remaining adsorption potential gradually weakens, thereby leading to a decrease in the adsorption capacity for As(V). It is worth noting that soil microaggregates with a particle size of (<0.002 mm) show the most significant decline. This is mainly because the smaller the soil particle size, the more adsorption sites it provides. In the initial stage of adsorption, As(V) rapidly binds to those sites in the soil with high binding energy, causing a sharp increase in adsorption capacity and significantly reducing the remaining adsorption potential. Ultimately, when adsorption equilibrium is reached, the differences in adsorption potential among microaggregates of various particle sizes become relatively small. The Freundlich equation provided a better fit (higher R^2 values) to the experimental data compared to the Langmuir equation (Table 2). This suggests that the Freundlich model, as an empirical equation, more effectively describes the adsorption behavior of As(V) onto these complex soil aggregates under the studied conditions. The Langmuir model's assumption of homogeneous monolayer adsorption may not fully capture the interactions in this heterogeneous soil system. Moreover, during the adsorption process, complex mechanisms such as surface precipitation or pore filling may occur.

The desorption property of black soil microaggregates shows a trend of rapid increase first and then slow growth within different particle size ranges. This phenomenon is related to the basic physical and chemical properties of the soil. Specifically, the desorption capacity of soil for As(V) is negatively correlated with the content of soil components. An increase in the content of SOM, Fe_d , and CEC will lead to an increase in the specific adsorption rate, making it more difficult for As(V) adsorbed in the soil to be desorbed, while the non-specific adsorption rate shows a downward trend [32]. This is mainly because iron and aluminum oxides are rich in variable positive and negative charges, which have a significant adsorption effect on heavy metal ions and oxygen-containing acid radical ions. Therefore, when the content of iron and aluminum oxides in the soil increases, the

specific adsorption capacity is also enhanced, making the desorption of As(V) in the soil more difficult.

Compared with the soil components that were not removed, the adsorption capacity of black soil for As(V) was significantly reduced after the removal of SOM and Fe_d components. However, statistical analysis (one-way ANOVA followed by Tukey's HSD test at $p < 0.05$) revealed that when the initial concentration of As(V) was low, there was no statistically significant difference in the adsorption capacity of black soil for As(V) before and after the removal of SOM or Fe_d . This might be because the initial concentration of As(V) is relatively low. Even if the organic matter in the soil is removed, the inorganic colloids in the soil still have a certain adsorption capacity, which is far from their saturated adsorption capacity. With the increase in the initial concentration of As(V), the adsorption of inorganic colloids in the soil approached saturation after the removal of components. However, organic colloids contain different oxygen-containing functional groups, which can undergo complexation or chelation reactions with As(V) and have a strong competitive adsorption capacity for As(V). Therefore, their adsorption capacity is larger than that of microaggregates after the removal of SOM. Fe_d carries a net positive charge under acidic conditions, thereby increasing the overall positive surface charge of the soil. The higher the content of Fe_d , the more conducive it is to the adsorption of As(V) by the soil. Therefore, the adsorption capacity of the original soil is higher than that of the microaggregates after removing Fe_d . After the removal of soil components, the easy desorption rate of black soil rises sharply. This is mainly because removing SOM and Fe_d eliminates the key sources of specific adsorption sites. Fe_d oxides provide surface hydroxyl groups (-OH), which exchange ligands with As(V) oxygen-containing anions to form inner-sphere complexes [35]. SOM contains oxygen-containing functional groups and can form strong complexes with metals/metalloids [40]. The removal of soil coordination groups significantly reduces the soil's ability to respond to this specific (chemical adsorption) interaction, resulting in a relative increase in the proportion of weak, non-specific adsorption [17].

5. Conclusions

The key factors controlling the adsorption and desorption of As(V) in black soil microaggregates were systematically revealed, and the differential effects of particle size classification and core components (SOM and Fe_d) were clarified. The adsorption capacity increases significantly with the decrease in particle size. Although the (<0.002 mm) component has an extremely low mass proportion, it becomes the main enrichment carrier of As in black soil due to having the largest specific surface area and the highest SOM and Fe_d content and shows the lowest desorption rate. SOM and Fe_d have been shown to be the leading contributors to specific adsorption (fixation), and they promote the stable binding of As(V) in the form of a monodentate or bidentate inner-ring complex by providing surface hydroxyl and oxygen-containing functional groups. After the removal of these components, the proportion of non-specific adsorption increased significantly, and the potential migration and environmental risk of As increased. The Freundlich model provided a better fit to the adsorption data than the Langmuir model under the studied conditions. K and Q_m were strongly positively correlated with SOM, Fe_d , and CEC contents. These findings elucidated the retention and stabilization behavior of As in black soil at the microaggregate scale and revealed the core role of fine particles and their active component complexes in As storage. At the practical level, remediation strategies for As-contaminated black soil should prioritize stabilizing these fine particles and highly reactive components. Future studies need to incorporate in situ spectroscopy to directly characterize the As morphology at the active interface and verify the relevant mechanism under field conditions.

Author Contributions: Conceptualization, L.H. and J.L.; methodology, P.Z.; software, J.L. and Q.Z.; validation, R.Y., S.T. and T.W.; formal analysis, L.H.; investigation, Q.Z.; resources, G.Y. and H.G.; data curation, P.Z. and S.T.; writing—original draft preparation, J.L.; writing—review and editing, G.Y.; visualization, R.Y.; supervision, T.W.; project administration, R.Y.; funding acquisition, L.H. and H.G. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by the National Natural Science Foundation of China (41807130); Special Fund for Science and Technology Innovation Teams of Shanxi Province (202304051001016); the Central Guidance on Local Science and Technology Development Fund (YDZJSX2024D051); Project of the Yellow River Midstream Ecological Protection and Restoration Engineering Technology Innovation Center of the Ministry of Natural Resources (HHZY-2025-003); and the Shanxi Province Industry-Education Integration Graduate Joint Training Base (2024JD11).

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: The data are available on request from the authors. The data are not publicly available due to confidentiality agreements.

Conflicts of Interest: Author Qian Zhang was employed by the company Standard Technology Engineering (Qing Dao) Co., Ltd. The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

References

1. Bhattacharya, P.; Welch, A.H.; Stollenwerk, K.G.; Mclaughlin, M.J.; Bundschuh, J.; Panaullah, G. Arsenic in the environment: Biology and Chemistry. *Sci. Total Environ.* **2007**, *379*, 109–120. [\[CrossRef\]](#)
2. Huo, L.; Huang, D.; Zeng, X.; Su, S.; Wang, Y.; Bai, L.; Wu, C. Arsenic availability and uptake by edible rape (*Brassica campestris* L.) grown in contaminated soils spiked with carboxymethyl cellulose-stabilized ferrihydrite nanoparticles. *Environ. Sci. Pollut. Res.* **2018**, *25*, 15080–15088. [\[CrossRef\]](#) [\[PubMed\]](#)
3. Li, Y.; Ji, L.; Mi, W.; Xie, S.; Bi, Y. Health risks from groundwater arsenic on residents in northern China coal-rich regio. *Sci. Total Environ.* **2021**, *773*, 145003. [\[CrossRef\]](#)
4. Li, Y.; Zhang, M.; Mi, W.; Ji, L.; He, Q.; Xie, S.; Xiao, C.; Bi, Y. Spatial distribution of groundwater fluoride and arsenic and its related disease in typical drinking endemic region. *Sci. Total Environ.* **2024**, *906*, 167716. [\[CrossRef\]](#) [\[PubMed\]](#)
5. Kumpiene, J.; Fitts, J.P.; Mench, M. Arsenic fractionation in mine spoils 10 years after aided phytostabilization. *Environ. Pollut.* **2012**, *166*, 82–88. [\[CrossRef\]](#)
6. Li, S.; Jia, M.; Li, Z.; Ke, X.; Wu, L.; Christie, P. Ecotoxicity of arsenic contamination toward the soil enchytraeid *Enchytraeus crypticus* at different biological levels: Laboratory studies. *Ecotoxicol. Environ. Saf.* **2021**, *207*, 111218. [\[CrossRef\]](#)
7. Xing, B.-S.; Liu, J.-D.; Liu, X.-B.; Han, X.-Z. Extraction and characterization of humic acids and humin fractions from a black soil of China. *Pedosphere* **2005**, *15*, 1–8.
8. Guo, G.L.; Zhou, Q.X. Contaminative trends of heavy metals in phaeozem of northeast China. *Univ. Chin. Acad. Sci.* **2004**, *21*, 386–392.
9. Bai, X.H.; Yan, H.M.; Pan, L.H.; Huang, H.Q. Multi-Agent Modeling and Simulation of Farmland Use Change in a Farming–Pastoral Zone: A Case Study of Qianjingou Town in Inner Mongolia, China. *Sustainability* **2015**, *7*, 14802–14833. [\[CrossRef\]](#)
10. Yang, N.; Winkel, L.; Johannesson, K. Predicting Geogenic Arsenic Contamination in Shallow Groundwater of South Louisiana, United States. *Environ. Sci. Technol.* **2014**, *48*, 5660–5666. [\[CrossRef\]](#) [\[PubMed\]](#)
11. Shrivastava, A.; Barla, A.; Singh, S.; Mandraha, S.; Bose, S. Arsenic contamination in agricultural soils of Bengal deltaic region of West Bengal and its higher assimilation in monsoon rice. *Hazard. Mater.* **2017**, *324*, 526–534. [\[CrossRef\]](#)
12. Feitosa, M.M.; Alvarenga, I.F.S.; Jara, M.S.; de Oliveira Lima, G.J.E.; Vilela, F.J.; Resende, T.; Guilherme, L.R.G. Environmental and human-health risks of as in soils with abnormal arsenic levels located in irrigated agricultural areas of Paracatu (MG), Brazil. *Ecotoxicol. Environ. Saf.* **2021**, *226*, 112869. [\[CrossRef\]](#) [\[PubMed\]](#)
13. Ljung, K.; Selinus, O.; Otabbong, E.; Berglund, M. Metal and arsenic distribution in soil particle sizes relevant to soil ingestion by children. *Appl. Geochem.* **2006**, *21*, 1613–1624. [\[CrossRef\]](#)
14. Jiang, W.; Zhang, S.; Shan, X.-Q.; Feng, M.; Zhu, Y.-G.; McLaren, R.G. Adsorption of arsenate on soils. Part 2: Modeling the relationship between adsorption capacity and soil physiochemical properties using 16 Chinese soils. *Environ. Pollut.* **2005**, *138*, 285–289. [\[CrossRef\]](#) [\[PubMed\]](#)

15. Tani, Y.; Miyata, N.; Ohashi, M.; Ohnuki, T.; Seyama, H.; Iwahori, K.; Soma, M. Interaction of Inorganic Arsenic with Biogenic Manganese Oxide Produced by a Mn-Oxidizing Fungus, Strain KR21-2. *Environ. Sci. Technol.* **2004**, *38*, 6618–6624. [\[CrossRef\]](#)
16. Khanam, R.; Kumar, A.; Nayak, A.K.; Shahid, M.; Tripathi, R.; Vijayakumar, S.; Bhaduri, D.; Kumar, U.; Mohanty, S.; Panneerselvam, P.; et al. Metal(loid)s (As, Hg, Se, Pb and Cd) in paddy soil: Bioavailability and potential risk to human health. *Sci. Total Environ.* **2020**, *699*, 134330. [\[CrossRef\]](#)
17. Weng, L.; Van Riemsdijk, W.H.; Hiemstra, T. Effects of Fulvic and Humic Acids on Arsenate Adsorption to Goethite: Experiments and Modeling. *Environ. Sci. Technol.* **2009**, *43*, 7198–7204. [\[CrossRef\]](#)
18. Ding, Z.; Hu, S.; Zhu, L.; Xiao, J.; Ye, Q.; Liu, T.; Shi, Z. Multiple effects of iron oxides on the adsorption and oxidation of dissolved organic matter by manganese oxides. *Geochim. Cosmochim. Acta* **2024**, *384*, 213–227. [\[CrossRef\]](#)
19. Hua, C.; Ma, W.; Li, J.; Wang, Y.; Zheng, F.; Zhang, Y.; Ren, Y.; Chen, T.; Li, H.; Bianchi, F.; et al. Internal Exposure Risks of Particle-Bound Polycyclic Aromatic Hydrocarbons with an Hourly Time Resolution to Humans in Beijing. *Environ. Sci. Technol.* **2025**, *59*, 12035–12047. [\[CrossRef\]](#)
20. Wang, Y.; Zhang, G.; Zhang, F.; Wang, H. Diagnostic strategy for the combined effects of microplastics and potentially toxic elements on microbial communities in catchment soil. *Sci. Total Environ.* **2023**, *860*, 160499. [\[CrossRef\]](#)
21. Peng, X.; Yan, X.; Zhou, H.; Zhang, Y.Z.; Sun, H. Assessing the contributions of sesquioxides and soil organic matter to aggregation in an Ultisol under long-term fertilization. *Soil Tillage Res.* **2015**, *146*, 89–98. [\[CrossRef\]](#)
22. Yin, Y.; Liang, C.-H.; Pei, Z.-J. Effect of greenhouse soil management on soil aggregation and organic matter in northeast China. *Catena* **2015**, *133*, 412–419. [\[CrossRef\]](#)
23. Zhang, Q.; Zhou, W.; Liang, G.; Sun, J.; Wang, X.; He, P. Distribution of soil nutrients, extracellular enzyme activities and microbial communities across particle-size fractions in a long-term fertilizer experiment. *Appl. Soil Ecol.* **2015**, *94*, 59–71. [\[CrossRef\]](#)
24. Xing, X.; Li, Z.; Wang, Y.; Tian, Z.; Cheng, J.; Hao, Z. Exploration of the interaction mechanism in the synergistic degradation of benzene and toluene over MnCoO_x catalysts. *Front. Environ. Sci. Eng.* **2025**, *19*, 22. [\[CrossRef\]](#)
25. Zhang, G.X.; Ren, R.; Yan, X.R.; Zhu, Y.E.; Zhang, H.Y.; Yan, G.Y. The key role of magnetic iron-to-biochar mass ratios in the dissipation of oxytetracycline and its resistance genes in soils with and without biodegradable microplastics. *Environ. Manag.* **2025**, *377*, 124658. [\[CrossRef\]](#) [\[PubMed\]](#)
26. Chang, J.N.; Zhang, H.B.; Liang, J.S.; Zhang, G.M.; Zhang, Y.J.; Zhang, R.; Wang, M.Q.; Lam, S.S.; Guo, S.Q.; Zhang, P.Y. Aging enhancement and synergistic effect on toxicity to soil fauna by polystyrene microplastics-tetrabromobisphenol A toxicity exposure. *Environ. Res.* **2026**, *288*, 123306. [\[CrossRef\]](#) [\[PubMed\]](#)
27. Gong, Y.; Wang, M.; Tian, S. Stability and Transport of Ferrihydrite Nanoparticles in Alkaline Soils. *Water Air Soil Pollut.* **2026**, *237*, 301. [\[CrossRef\]](#)
28. Cui, H.; Ma, K.; Fan, Y.; Peng, X.; Mao, J.; Zhou, D.; Zhang, Z.; Zhou, J. Stability and heavy metal distribution of soil aggregates affected by application of apatite, lime, and charcoal. *Environ. Sci. Pollut. Res.* **2016**, *23*, 10808–10817. [\[CrossRef\]](#)
29. Li, S.; Wang, M.; Zhao, Z.; Li, X.; Han, Y.; Chen, S. Alleviation of cadmium phytotoxicity to wheat is associated with Cd re-distribution in soil aggregates as affected by amendments. *RSC Adv.* **2018**, *8*, 17426–17434. [\[CrossRef\]](#)
30. Zhang, T.; Jiku, M.S.; Li, L.; Ren, Y.; Li, L.; Zeng, X.; Colinet, G.; Sun, Y.; Huo, L.; Su, S. Soil ridging combined with biochar or calcium-magnesium-phosphorus fertilizer application: Enhanced interaction with Ca, Fe and Mn in new soil habitat reduces uptake of As and Cd in rice. *Environ. Pollut.* **2023**, *332*, 121968. [\[CrossRef\]](#)
31. Lombi, E.; Sletten, R.S.; Wenzel, W.W. Sequentially extracted Arsenic from different size fractions of contaminated soils. *Water Air Soil Pollut.* **2000**, *124*, 319–332. [\[CrossRef\]](#)
32. Lu, R.K. *Analytical Methods for Soil and Agro-Chemistry*, 1st ed.; China Agricultural Science and Technology Press: Beijing, China, 2000; pp. 1–10.
33. Li, Z.W.; Huang, B.; Huang, J.Q. Influence of removal of organic matter and iron and manganese oxides on cadmium adsorption by red paddy soil aggregates. *RSC Adv.* **2015**, *5*, 90588–90595. [\[CrossRef\]](#)
34. Six, J.; Paustian, K.; Elliott, E.T.; Combrink, C. Soil Structure and Organic Matter I. Distribution of Aggregate-Size Classes and Aggregate-Associated Carbon. *Soil Sci. Soc. Am.* **2000**, *64*, 681–689. [\[CrossRef\]](#)
35. Sparks, D.L. *Chapter 7: Sorption Phenomena. Environmental Soil Chemistry*, 2nd ed.; Academic Press: Cambridge, MA, USA, 2003.
36. Limousin, G.; Gaudet, J.P.; Charlet, L.; Szenknect, S.; Barthès, V.; Krimissa, M. Sorption isotherms: A review on physical bases, modeling and measurement. *Appl. Geochem.* **2007**, *22*, 249–275. [\[CrossRef\]](#)
37. Covelo, E.F.; Vega, F.A.; Andrade, M.L. Sorption and desorption of Cd, Cr, Cu, Ni, Pb and Zn by a Fibric Histosol and its organo-mineral fraction. *Hazard. Mater.* **2008**, *159*, 342–347. [\[CrossRef\]](#)
38. Wenzel, W.W.; Kirchbaumer, N.; Prohaska, T.; Stingeder, G.; Lombi, E.; Adriano, D.C. Arsenic fractionation in soils using an improved sequential extraction procedure. *Anal. Chim. Acta* **2001**, *436*, 309–323. [\[CrossRef\]](#)
39. Sherman, D.M.; Randall, S.R. Surface complexation of arsenic(V) to iron (III) (hydr)oxides: Structural mechanism from ab initio molecular geometries and EXAFS spectroscopy. *Geochim. Cosmochim. Acta* **2003**, *67*, 4223–4230. [\[CrossRef\]](#)

40. Goldberg, S.; Johnston, C.T. Mechanisms of Arsenic Adsorption on Amorphous Oxides Evaluated Using Macroscopic Measurements, Vibrational Spectroscopy, and Surface Complexation Modeling. *Colloid Interface Sci.* **2001**, *234*, 204–216. [[CrossRef](#)]
41. Sabine, G. Competitive Adsorption of Arsenate and Arsenite on Oxides and Clay Minerals. *Soil Sci. Soc. Am.* **2002**, *66*, 413–421.
42. Dixit, S.; Hering, J.G. Comparison of arsenic(V) and arsenic (III) sorption onto iron oxide minerals: Implications for arsenic mobility. *Environ. Sci. Technol.* **2003**, *37*, 4182–4189. [[CrossRef](#)]
43. Redman, A.D.; Macalady, D.L.; Ahmann, D. Natural organic matter affects arsenic speciation and sorption on hematite. *Environ. Ment. Sci. Technol.* **2002**, *36*, 2889–2896. [[CrossRef](#)] [[PubMed](#)]

Disclaimer/Publisher’s Note: The statements, opinions and data contained in all publications are solely those of the individual author(s) and contributor(s) and not of MDPI and/or the editor(s). MDPI and/or the editor(s) disclaim responsibility for any injury to people or property resulting from any ideas, methods, instructions or products referred to in the content.