Comparison on Copper (II) Desorption from Loess Soil with and without Biochars Derived from Flax and Rape Straws

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ABSTRACT: It has been reported that biochar is capable of adsorbing and immobilizing heavy metals in soils due to its prominent properties, such as the porous structure, large and charged surface, and functional groups on surface. The desorption of Cu (II) (Cu) from loess soil was studied and compared using batch washing method, with and without biochars derived from flax and rape straws. The effects of contact time, initial pH value of solution, temperature, initial concentration of Cu in soil, and co-existing Ca\(^{2+}\) on desorptions were investigated. The pseudo-second-order kinetic model could be used to describe the desorbing process well. The leaching ability (\(\eta\)) of Cu from solid phase increased in overly acidic solutions. Increasing temperature improved desorption of Cu from loess soil. When the initial concentration of Cu in soil increased, the \(\eta\) values of Cu decreased. Slight effect of Ca\(^{2+}\) on the \(\eta\) value of Cu was found. No significant immobilization of Cu by biochars was found because loess soil is much capable of holding Cu, possibly due to high content of carbonates. The results could provide references for immobilization remediation of loess soils contaminated with heavy metals using biochars as amendment.

INTRODUCTION

In recent years, increasingly more soils are found to be contaminated with heavy metals due to waste emissions from industrial production, mining activities, waste application and wastewater irrigation, which has brought serious environmental problems [1–3]. For example, the arable land area subjected to heavy metal contamination is about 20 million ha, accounting for 20% of the total agricultural land area in China [4]. Heavy metals in soils are not only harmful to ecosystem and agricultural production but also serious threat to human being [5,6]. Loess, a special soil spreading over the large area of northwestern China and central Asia, being different from the acidic soils due to its high pH value, high calcareous content, loose structure, large porosity and water permeability, and low agglomerating force and organic carbon content, is also found to have suffered much heavy metal contamination due to the mining activity and sewage irrigation [7–9]. Therefore, it was quite necessary for the research on heavy metal pollution in loess, and on how to effectively control and manage pollution, which has gained the attention of the government and scholars. At the present time, biochar is a promising amendment into soils, improving soil properties and remediating the contaminated soils [10,11].

Biochar is produced by so-called thermal decomposition of organic materials (wood, straw, leaves, manure and sludge etc.) under limited supply of oxygen (O\(_2\)) and at relatively low temperatures (< 700°C). Biochar is carbonaceous with microporous structure, active surface charge and functional groups [12–14]. These properties have sparked strong interesting in mitigating climate change, improving soils, enhancing crop production and controlling contamination in soils [15]. Although the effect of biochar on heavy metal mobility and bioavailability in soils has been investigated, the results may be quite different when the biochars derived from various bomasses are used. Biochar addition led to immobilization of both Cd and Zn in soil in a column leaching experiment [16]. Consequently, Cd and Zn concentrations in pore water were reduced 300 and 45-fold, respectively [16]. Namgay et al. reported that the concentrations of extractable As and Zn in soil increased with biochar addition, whereas the concentration of extractable Pb decreased, that of Cu did not change, and that of Cd showed an inconsistent trend [17]. Heavy metal ions (Cu) were tested for their mobility in San Joaquin soil.
amended with 5–20% (w/w) biochars, which resulted in negligible remediation in soluble Cu concentrations [18]. Cu contamination in soils is very common in the research area. Baiyin for example being 80 km from the research site is an important industrial base of copper metal. Because of the history of metal smelting and sewage irrigation, the average concentration of heavy metals in farmland is high, and the average concentration of Cu (II) is 90.12 mg·kg⁻¹, which is much higher than that of background in soil [19]. In the previous studies, researchers have mainly focused on the acidic type of soils amended with biochar derived from the local biomasses. However, there are few studies on the effect of biochars on heavy metal (e. g. Cu) migration and conversion in loess soils, to our knowledge.

In this study, the flax (Linum usitatissimum L.) straw and rape (Brassica campestris L.) straw, which are widely planted and abandoned in loess areas, were pyrolyzed into biochars at 600°C. A comparison was conducted on the effects of biochars on copper (II) (Cu) desorption from loess soil using batch washing method. The factors influencing the leaching ability (η) of Cu such as contact time, initial pH value of solution, initial Cu concentration in soil, temperature, and co-existing cation (Ca²⁺) were tested. The objective is to investigate preliminarily whether the immobilization of heavy metals occurs and the pattern of heavy metal desorption changes in loess soil-water system in the presence of biochar.

**EXPERIMENTAL SECTION**

**Chemicals**

Copper nitrate (Cu(NO₃)₂·3H₂O), sodium nitrate (Na(NO₃)₂), calcium nitrate (Ca(NO₃)₂), sodium hydroxide (NaOH), and nitric acid (HNO₃) were purchased from Guangfu Chemical Research Institution, China. All reagents used were of analytical grade. The deionized water was used for all procedures.

**Materials**

The loess soil (light sierozem) sample (0–20 cm) was collected at a hill in Lanzhou Jiaotong University, China, where few low artemisia plants grew. The soil was air dried, removed weeds and gravels, and passed through a 40 mesh sieve. The pH value of the soil was determined as 8.11 [1:2.5 of soil mass (g) to volume of water (mL)] on a pH meter (PHS-3C, Electronics and Scientific Instrument Corporation, Shanghai, China).

The background content of Cu is 22.5 mg kg⁻¹ [20]. Its value of the point of zero charge (pH<sub>PZC</sub>) is 2.82 [21] and the carbonate content is 3.9% [22]. The loess organic matter was determined as 0.53% using digestion method by potassium dichromate. The soil contains 18.65% grave, 28.50% silt and 52.85% clay. The loess was artificially prepared as Cu-contaminated soil with Cu(NO₃)₂·3H₂O solution. Desired amounts of Cu(NO₃)₂ was dissolved in deionized water and added to the contaminant-free soils. The slurry mixtures were evaporated under a hood, and the soils spiked with Cu were mixed, homogenized, passed through a 40 mesh sieve and aged for 20 days in order to achieve a relative stable distribution of species fraction of Cu. The final Cu concentrations in the loess soils were 400, 600, 700, 1000 and 1500 mg kg⁻¹, respectively.

The used biochars derived from flax and rape straw were prepared and characterized through the same processes as reported in our previous study [23]. In brief, the cleaned flax straw and rape straw chips were pyrolyzed in muffle oven at temperature 600°C for 4 h. The results from pre-experiment showed that the biochars obtained from 600°C and 4 h pyrolysis exhibited much adsorption capacities for Cu [23]. After cooling to room temperature, the resulting biochars were washed with 0.1 mol L⁻¹ HCl to remove excess ash [8], rinsed with deionized water, oven-dried for 12h at 80°C, and then sieved to 100 mesh. The biochars were termed as LS600 and BS600, respectively. The physical and chemical parameters of two biochars are listed in Table 1 [23], where the pH values and ash contents of the biochars were determined according to the national standards of China [24]. The point of zero charge (pH<sub>PZC</sub>) was determined by potentiometric titration. Elemental analysis was completed on an elemental analyzer (Vario EL, Germany). The biochar contents of C, N, H and S were determined directly, while the O content was calculated by subtracting the contents of C, N, H, S and the ash contents from the total mass of the sample. The specific surface areas of the biochars were determined on a surface and porosimetry analyzer (Micromeritics ASAP 2020, USA). The surface areas were measured with N₂ adsorption at 77 K using BET adsorption isotherms.

**Batch Washing Procedure**

A series of 0.2 g of the contaminated soil, 0.2 g of the contaminated soil with 0.05 g of LS600, or 0.2 g of the contaminated soil with 0.05 g of BS600 were added to 20 mL of 0.01mol L⁻¹ NaNO₃ solutions in 50-mL
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flasks with taps. 0.1 mL of 1 g L\(^{-1}\) NaN\(_3\) was added to inhibit microbial growth [25]. Duplicate samples were prepared. Then the samples were placed on a reciprocating shaker (THZ-82A, Jiangsu Quartz Glass Factory, China) at 150 rpm and 25°C. After shaking, each suspension was centrifuged at 3500 rpm for 30 min, and the supernatant was carefully decanted into a clear glass vial and filtered through a 0.45 μm membrane. The concentration of Cu in the filtrate was measured by atomic absorption spectrophotometry (AAS). The pH values of the solutions were adjusted to the target ones by 1 mol L\(^{-1}\) HNO\(_3\) and 1 mol L\(^{-1}\) NaOH solutions [8].

The contact time, initial pH value of solution, dosage of contaminated soil, temperature, initial Cu concentration in soil and electrolyte concentration were kept as 24 h, 7.0, 0.2 g, 25°C, 600 mg Cu kg\(^{-1}\) soil, and 0.01 mol L\(^{-1}\) NaNO\(_3\) respectively except that one of the factors was tested and changed.

### Analytical Method

The concentration of Cu in aqueous phase was determined on an atomic absorbance spectrophotometer (Varian, Spectrum AA110/220, USA) at 324.8 nm. The flame type was air-acetylene one. Standard solutions were prepared and used to automatically draw the standard calibration. The leaching ability (\(\eta, \%\)) of Cu can be calculated as follows in Equation (1) [26]:

\[
\eta = \frac{C_e V_e}{C m} \times 100\%
\]

where \(C_e\) is the Cu concentration in elution (mg L\(^{-1}\)), \(V_e\) is the volume of elution (mL), \(C\) is the initial Cu concentration in soil (mg kg\(^{-1}\)), and \(m\) is the weight of soil (g). All of the values of \(\eta\) were the means of parallel determination.

### RESULTS AND DISCUSSION

#### Effect of Contact Time

Figure 1 presents the plots of \(\eta\) values versus the contact time varying from 0 to 24 h. It is indicated that the removal of Cu from solid phase increased as time increasing but the removal amount was not large and less than 3%. Within 1 h, sharp enhancements of \(\eta\) were observed in the three cases. The rates increased slow with time ranging from 1 to 6 h and then leveled off after 6 h [27]. For the single loess system, it was found that a strong immobilizing ability of loess for Cu occurred because only up to 2.8% of the total amount of Cu was desorbed into aqueous phase. In our previous study, it was found that the adsorptive capacity of loess soil for Cu could reach 16.13 g kg\(^{-1}\) [28]. This is attributed to the high content of carbonates in loess soil [29,30]. It was demonstrated that binding of Cu(II) bonded to carbonates is dominant in the copper sorption by loess soils [29]. Zhao et al. pointed out that the direct precipitation and co-precipitation of Cu (II) with carbonates in loess soils were the main mechanisms on Cu (II) sorption onto loess soils, which resulted in a large fraction of carbonate-bound copper and a large holding capacity of loess for copper [22]. For example, the capacity of 5.5 g of Malan loess for copper (II) was still unsaturated even while the initial concentration of copper in aqueous solution was up to 957.0 mg L\(^{-1}\). The sorption of copper decreased significantly after extrac-

### Table 1. Physical and Chemical Parameters of Biochars.

<table>
<thead>
<tr>
<th>Biochar</th>
<th>pH</th>
<th>pH(_{pec})</th>
<th>Ash, %</th>
<th>BET Surface Area, m(^2) g(^{-1})</th>
<th>Element Content, wt%</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>C</td>
<td>H</td>
<td>N</td>
<td>S</td>
<td>O</td>
</tr>
<tr>
<td>BS600</td>
<td>9.98</td>
<td>5.08</td>
<td>29.2</td>
<td>158</td>
<td>53.6  1.56  0.86  0.76  14.0</td>
</tr>
<tr>
<td>LS600</td>
<td>10.7</td>
<td>4.73</td>
<td>19.1</td>
<td>109</td>
<td>60.1  1.80  14.83  0.61  16.5</td>
</tr>
</tbody>
</table>
tion of carbonates [22]. The possible precipitates of Cu on loess soils, e.g. Cu$_3$(OH)$_2$(CO$_3$)$_2$ and Cu$_2$(OH)$_2$CO$_3$, are extremely undissolved, e.g. Cu$_3$(OH)$_2$(CO$_3$)$_2$ $pK_{sp} = 21.53$ and Cu$_2$(OH)$_2$CO$_3$ $pK_{sp} = 21.55$ [22]. Thus, ultralow concentrations of Cu in aqueous phase would be found.

As for the systems of loess with BS600 and LS600, the whole patterns of desorption processes were similar to that of the single loess system. However, the Cu leaching abilities decreased by 1.9% and 1.8% at large. This indicated that the immobilization of Cu was somewhat enhanced in the presence of used biochars. No doubt, the additional immobilized amount of Cu was due to the adsorptive capacity of BS600 and LS600 [23] and the negligible difference of the Cu leaching abilities between in the presence of BS600 and LS600 might be due to their similar adsorptive capabilities for Cu [23], because their structure and surface properties are not significantly different (See Table 1) [23]. Frišták et al. reported Cu$^{2+}$ ions adsorption onto the biochars derived from two woody biomasses at the same pyrolysis conditions [31]. It was found that the adsorption of Cu$^{2+}$ onto the biochars followed pseudo-second-order kinetic model, Langmuir and Freundlich isotherm model and it was confirmed that the crucial roles in adsorption processes were mainly hydroxyl and carboxyl functional groups on the biochars’ surface by chemical modification (esterification of carboxyl, and methylation of hydroxyl and phenolic functional groups of biochar) and FTIR analyses [31]. In our previous study [28], no significant difference occurred between the results of surface functional group determination for BS600 and LS600 using Boehm titration and the adsorption of Cu onto BS600 and LS600 followed pseudo-second-order kinetic model and Langmuir isotherm model. Therefore, although there was a large difference between the BET surface area of BS600 and LS600 (See Table 1), very similar values of the maximum adsorptive capacities of Cu (30.8 mg g$^{-1}$ for BS600 and 26.7 mg g$^{-1}$ for LS600) were found [28]. The values of standard Gibbs energy ($\Delta G^\circ$) for the adsorption of Cu onto loess, BS600 and LS600 were determined as $-13.87$ kJ mol$^{-1}$, $-8.78$ kJ mol$^{-1}$ and $-9.16$ kJ mol$^{-1}$ at 25°C [28], which indicate Cu ions are much easier to be adsorbed onto loess, rather than the used biochars. It can be deduced that it was impossible that most of Cu adsorbed on loess phase was transferred into biochar phase at the tested experimental conditions. Thus, BS600 and LS600 did not exhibit much immobilization results compared with the results mentioned above [16] although their adsorptive capacities for Cu (30.8 mg g$^{-1}$ and 26.7 mg g$^{-1}$) were large [23] and the used amount of biochars was up to 25% of the total solid weight. The main reason might be the prominent holding ability of loess soils to Cu, which led to ultralow concentrations of Cu in aqueous phase and thus the immobilization by BS600 and LS600 was not significant.

The desorption mechanism for Cu could be explained using pseudo-first-order kinetic and pseudo-second-order kinetic [26]. The model’s linear forms may be expressed as seen in Equations (2) and (3):

$$\log(q_e - q_t) = \log q_e - \frac{k_1}{2.303} t$$

$$(3) \quad \frac{t}{q_t} = \frac{1}{k_2q_e^*} + \frac{1}{q_e} t$$

where $q_e$ (mg g$^{-1}$) denotes the desorbed amount of metal per unit mass of solid at equilibrium; $q_t$ (mg g$^{-1}$) is the desorbed amount of metal per unit mass of solid at time $t$ (h); $k_1$ (h$^{-1}$) is the constant of pseudo-first-order kinetic model; $k_2$ (g mg$^{-1}$ h$^{-1}$) is the constant of pseudo-second-order kinetic one. According to data seen in Figure 1 and Equations (2) and (3), regression results were obtained and listed in Table 2. It is apparent the pseudo-second-order kinetic model was better used to describe the desorption processes than the pseudo-first-order kinetic one due to high values of correlation coefficients ($R^2$), 0.9960, 0.9960 and 0.9980 for single loess, loess with BS600 and loess with LS600 systems, together with the slight difference between the desorbed amount of Cu tested at equilibrium ($q_{e,ex}$, mg g$^{-1}$) and calculated from regression ($q_{e,cal}$, mg g$^{-1}$). The

<table>
<thead>
<tr>
<th>System</th>
<th>$q_{e,ex}$ mg g$^{-1}$</th>
<th>$q_{e,cal}$ mg g$^{-1}$</th>
<th>$k_1$, h$^{-1}$</th>
<th>$R^2$</th>
<th>$q_{e,cal}$ mg g$^{-1}$</th>
<th>$k_1$, h$^{-1}$</th>
<th>$R^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Loess</td>
<td>0.0168</td>
<td>0.0069</td>
<td>0.168</td>
<td>0.857</td>
<td>0.0170</td>
<td>66.35</td>
<td>0.996</td>
</tr>
<tr>
<td>Loess + BS600</td>
<td>0.0123</td>
<td>0.0035</td>
<td>0.147</td>
<td>0.860</td>
<td>0.0124</td>
<td>137.0</td>
<td>0.996</td>
</tr>
<tr>
<td>Loess + LS600</td>
<td>0.0114</td>
<td>0.0025</td>
<td>0.122</td>
<td>0.952</td>
<td>0.0114</td>
<td>198.5</td>
<td>0.998</td>
</tr>
</tbody>
</table>

**Table 2. Regression Results for Kinetic Models.**
values of $q_{e,ex}$ in the three cases were 0.017, 0.0124 and 0.0114 mg g$^{-1}$, which were very close to those values of $q_{e,cal}$ from the regression results for pseudo-second-order kinetic model.

**Effect of pH Value**

Figure 2 shows the relationship between the leaching ability ($\eta$) of Cu and the initial pH value of solution, with pH values ranging from 2 to 6. As pH values located from 6 to 4, no obvious change in the leaching ability occurred in each system. The variation trends of $\eta$ in three cases were somewhat different when the pH values decreased, especially in overly acidic solutions. For the single loess system, when the pH values were less than 3.5, obvious increment in the values of $\eta$ was observed. Firstly, the reason might be the dissolution of carbonate precipitates of Cu on loess soil and more Cu ions were released into aqueous phase [32]. Secondly, when the pH values were less than pH$_{ZPC}$ 2.82, the dominant charge on loess soil was positive and the sorption of Cu ions was inhibited. Thirdly, more hydrogen ions in acidic solution would compete the adsorption of Cu ions onto soils. The Cu desorptive efficiencies from loess increased up to 4.95%, 4.24%, and 3.83% while the initial pH values of solutions were 2.0, 2.5 and 3.0, respectively.

As for the loess with biochar systems, when the pH values decreased from 4.0 to 2.5, slight increments in the leaching ability ($\eta$) of Cu were obtained. It is no doubt that dissolution of copper carbonates began in this pH value range. On paper, the Cu concentrations in aqueous phase would increase at a rate as that in the single loess system. However, it was the adsorption of Cu onto biochars that slowed down the rate. It is indicated that the predominance of Cu$^{2+}$ ions in solution is shown for the pH range less than 7.0 in water-biochar system [31]. Our previous study indicated that the mechanisms on adsorption of Cu on BS600 and LS600 were ion exchange and coordination of Cu with biochar surface [23]. The results reported by Uchimiya et al. suggested that higher pyrolysis temperature lead to the disappearance (e.g., aliphatic –CH$_2$ and –CH$_3$) and the formation (e.g., C–O) of certain surface functional groups of biochars [33]. Moreover, with higher carbonized fractions and loading of chars, Cu immobilization by cation exchange becomes increasingly outweighed by other controlling factors such as the coordination by n electrons (C＝C) of carbon and precipitation [33]. On the one hand, as shown in Table 1, the values of pH$_{ZPC}$ for BS600 and LS600 are 5.08 and 4.73, which demonstrates that positive charge occurred on the biochar surface and the adsorptive capacity of biochars for Cu decreased when the pH values were located in this range. On the other hand, the dissociation degree of oxygen-containing functional groups binding Cu such as –COOH and –OH [23] would decrease, which also led to reduction in adsorption ability of biochars [15,34]. The leaching ability ($\eta$) of Cu increased at a slow rate. After all, the positive charge and the acidic functional groups would be dominant when the solutions were much too acidic (pH < 2.5). Meanwhile, the competition of free protons and metal ions for unoccupied binding sites with lowered solution pH increased, which also caused the reduction of adsorption effectiveness for bivalent metal ions [31]. Then the leaching ability of Cu increased sharply, with 4.70% and 2.95% values in the presence of BS600 and LS600, respectively.

**Effect of Temperature**

Figure 3 illustrates the variation of $\eta$ values with temperature in three systems. It is showed that the desorptive rate of Cu increased continuously in the single loess system when the tested temperature increased from 25–40°C. Although the adsorption of Cu onto loess soil is endothermic [28], increasing of temperature will result in the water solubilities of carbonates on loess soil [35]. It is possible the latter action was dominant in the desorption process in the single loess system. As the systems with biochars were concerned, the desorptive rates of Cu did not obviously change when temperatures were set up as 25°C, 30°C, 35°C and 40°C. These results indicated that parts of
dissolved Cu were transferred and immobilized by the biochars through adsorption. Because the adsorption of Cu onto BS600 and LS600 was endothermic, the adsorptive capacity of the two biochars would increase with temperature increasing [28] which inhibited the Cu concentrations in aqueous phase.

Effect of Initial Cu Concentration

The effect of initial Cu concentration in loess soil on the leaching ability of Cu is seen in Figure 4. For all cases, the leaching ability of Cu decreased with the initial Cu concentrations increasing in loess soil. As 400 mg kg$^{-1}$ of the contaminated soil was used, the leaching ability of Cu was 3.55%, 3.00% and 2.80% in the single loess, loess with BS600 and loess with LS600 systems, respectively. While the initial Cu concentration in loess increased up to 1500 mg kg$^{-1}$, those values reduced to 0.78%, 0.61% and 0.6%, being low as about 5 times as those with 400 mg kg$^{-1}$ of the contaminated soil. As mentioned above, the concentration of Cu in aqueous phase depended mainly upon the weak dissolution of carbonates on loess soil and loess had a high adsorptive capacity for Cu [36]. The concentrations of Cu in aqueous phase changed slightly in all the cases. Therefore, the leaching ability of Cu decreased much because the denominator in Equation (1) increased rapidly when the initial Cu in soil was added too much.

Effect of Co-existing Cation (Ca$^{2+}$)

The effect of co-existing Ca$^{2+}$ on the leaching ability of Cu is illustrated in Table 3. It is observed that the desorption of Cu was not significantly changed when the concentration of Ca$^{2+}$ increased from 0–0.5 mmol L$^{-1}$ (0–20 mg L$^{-1}$). For the single loess system, slight decrease in the leaching ability of Cu was found. This could be attributed to the weak alkalinity of Ca(NO$_3$)$_2$ further reduced the solubility of carbonates. For the loess with biochar systems, a U shape of variation of the $\eta$ values was observed. On the one hand, the alkaline Ca(NO$_3$)$_2$ could reduce the solubility of carbonates, which led to the reduction in Cu concentrations in aqueous phase, as shown when Ca$^{2+}$ concentrations ranged from 0–0.03 mol L$^{-1}$. On the other hand, it was possible that much Ca$^{2+}$ would result in the desorption of Cu having been adsorbed on biochar due to the competitive adsorption, conversely, which resulted in increase of Cu in aqueous phase [37]. When the heavy metal ions enter the soil environment, the sorption mechanisms may include four types, i.e. (1) electrostatic action adsorption; (2) ion exchange; (3) complexing with organic and inorganic ligands; (4) chemical precipitation. If the sorption of a metal onto soil is based on one of the two formers, the adsorbed metal ions can usually be replaced in the neutral salt or buffer solution [38]. As discussed above, the main sorption

<table>
<thead>
<tr>
<th>System</th>
<th>Ca$^{2+}$, mmol L$^{-1}$</th>
<th>0</th>
<th>0.1</th>
<th>0.2</th>
<th>0.3</th>
<th>0.4</th>
<th>0.5</th>
</tr>
</thead>
<tbody>
<tr>
<td>Loess</td>
<td>2.62</td>
<td>2.74</td>
<td>2.45</td>
<td>2.13</td>
<td>2.04</td>
<td>2.02</td>
<td></td>
</tr>
<tr>
<td>Loess + BS600</td>
<td>1.77</td>
<td>1.64</td>
<td>1.40</td>
<td>1.28</td>
<td>1.41</td>
<td>1.57</td>
<td></td>
</tr>
<tr>
<td>Loess + LS600</td>
<td>1.50</td>
<td>1.19</td>
<td>1.07</td>
<td>1.04</td>
<td>1.50</td>
<td>1.44</td>
<td></td>
</tr>
</tbody>
</table>
mechanism of Cu onto loess is chemical precipitation. The cations of electrolytes are difficult in replacement of Cu in carbonates.

CONCLUSIONS

The desorptions of Cu in the loess and loess with biochars BS600 and LS600 could reach equilibrium at 24 h. The data of desorptions were well fitted by pseudo-second-order kinetic model. The leaching ability (η) of Cu from solid phase improved when the initial pH values of solution were less than 3.5 in loess system and 4.0 in loess with biochar systems. The values of η in the single loess system increased and those in the loess with biochar systems did not change with temperature increasing. When the initial concentration of Cu in soil increased, the η values of Cu decreased. No obvious effect of Ca²⁺ on the η value was found. The desorption patterns seemed similar among three cases but the η values of Cu in the presence of biochars were less than those in the absence of biochars. The used biochars, BS600 and LS600, did not exhibit much immobilization of Cu, possibly due to the strong holding capability of loess soil to Cu.

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