Transport of natural soil nanoparticles in saturated porous media: effects of pH and ionic strength

Yanji Jianga, Lin Yua, Huimin Suna, Xianqiang Yinb, Changzhao Wangb, Shiny Mathewsc and Nong Wanga

aCollege of Natural Resources and Environment, Northwest A&F University, Yangling, China; bShaanxi Entry-Exit Inspection and Quarantine Bureau, Xian, China; cSoil Science Department, North Dakota State University, Fargo, ND, USA; dAgro-Environmental Protection Institute, Ministry of Agriculture of the People’s Republic of China, Tianjin, China

ABSTRACT

To understand the effects of ionic strength and pH on the transport of natural soil nanoparticles (NS) in saturated porous media, aeolian sandy soil nanoparticles (AS), cultivated loessial soil nanoparticles (CS), manural loessial soil nanoparticles (MS) and red soil nanoparticles (RS) were leached with solutions of varying pH and ionic strength. The recovery rate of soil nanoparticles decreased in the order AS > RS > MS > CS. Transport of soil nanoparticles was enhanced with increasing pH and decreasing ionic strength and was attributable to changes in the Zeta potential of NS. Deposition of NS was also affected by the composition of soil nanoparticles and the surface charge. Column experiments showed that the interaction between soil nanoparticles and saturated quartz sand was mainly due to the physical and chemical properties of soil nanoparticles. The Derjaguin–Landau–Verwey–Overbeek interaction energies between NS and sand were affected by pHs and ionic strengths. Soil nanoparticles transport through saturated porous media could be accurately simulated by the one-dimensional advection-dispersion-reaction equation.

1. Introduction

Soil colloids generally have a diameter less than 10 μm and nanoparticles less than 100 nm [1]. Mobile soil colloids have a strong ability to adsorb contaminants and can migrate in porous media at a rate similar to the aqueous phase [2]. Due to the natural presence of nanomaterials in the environment [3] and the potential risks to the ecosystem and public health, pollutant transport of nanoparticles [4], colloids [5], and their associated contaminants [6] in saturated porous media has attracted more attention in the past decade. Advection, convection, precipitation and adsorption are a few of the physical and chemical processes that occur when colloids migrate in the porous media [7,8]. The adsorption of colloids in transport processes involves adhesion, deposition and release [9]. Fundamentally, the transport of colloidal particles in porous media depend on the colloidal size, shape and colloidal characteristics [10–12]. The smaller the size of the particles the easier it is for them to migrate through a porous media [13,14]. The transport of soil colloids in saturated porous media is also influenced by solution pH [15] and ionic strength [16]. In natural environments, both soil pH and ionic strength will change with variations in rainfall and irrigation [17]. The release of the colloids due to changes in pH and ionic strength can enhance the transport of environmental contaminants adsorbed to the colloidal surface [18]. Therefore a large number of studies the effects of pH and ionic strength on the transport of colloids in porous media are investigated [6,19] and used a single medium, quartz sand, to simulate soil medium [6,20].

In recent years, nanomaterials have been widely used and the transport of nanoparticles in the underground environment has become a worrying issue. They will enter in groundwater because of their tiny size, then they can adsorb contaminants due to their high specific surface area and carry these contaminants to move far apart [21,22]. The nanoparticles are usually better than or equal to the migration of the solute, the larger NPs are carried at a higher rate and are not close to the medium due to their size [23]. These factors explain why nanoparticles pose a significant risk to the subsurface environment. To date, most studies have focused on the fate and transport of artificial nanoparticles in the environment, such as titanium dioxide [24], graphene oxide [25] etc. A great deal of research has been done on the transport and retention of artificial nanoparticles under environmental conditions. For example, the effects of different physical and chemical conditions such as size, concentration, fluid velocity and solution chemistry (i.e. ionic...
strength, pH) [24,26,27] on the transport and retention of nanoparticles have been determined. These studies only focus on artificial nanomaterials, which really ignore the natural nanoparticles. Soil contains large amounts of nanoparticles, which is an important source of natural nanoparticles. However, there is no study of the fate and migration of soil nanoparticles.

There are significant differences in soil composition and soil composition, structural properties and organic matter content in different soil types in China. Soil types not only determine the migration of soil particles in the natural environment, but also affect the ability of heavy metals to transport in the soil [28]. Hence, unlike many studies on the transport of artificial nanoparticle or soil colloids in the porous media compacted with quartz sand, we studied the transport behaviors of natural soil nanoparticles (NS) by extraction method from four natural soil in China, aeolian sandy soil nanoparticles (AS), cultivated loessial soil nanoparticles (CS), manural loessial soil nanoparticles (MS) and red soil nanoparticles (RS). To better understand the transport of NS in a saturated porous media was examined and column leaching experiments were conducted at different with suspensions of NS. In addition, Derjaguin–Landau–Verwey–Overbeek (DLVO) theory was used to explain the behavior of NS transport in porous media under different pH and ionic strengths.

2. Materials and methods

2.1. Natural soil nanoparticles suspensions

Four different soil nanoparticles, aeolian soil nanoparticles (AS), cultivated loessial soil nanoparticles (CS), manural loessial soil nanoparticles (MS) and red soil nanoparticles (RS), were extracted from four different soils sampled from the agricultural fields at Shenmu Shaanxi, Ansai Ecological Experimental Station Shaanxi, Yanling Shaanxi and Panzhihui Sichuan, respectively.

Nanoparticle suspensions were derived from the parent soil following an aqueous extraction. Oven dried soil sample (20 g) was weighed into a beaker and distilled water (400 mL) was added. The sample was shaken at 170 rpm for 0.5 h in a constant temperature incubator shaker (HZQ-F100, Donglian Electron Technology Ltd., Harbin, China), followed by 30 min of ultrasonic dispersion in an ultrasonic cleaner (KQ-500DE, Kunshan Ultrasonic Instrument Company, China). The soil suspension was transferred to a measuring cylinder and allowed to settle for 24 h after which the soil suspension was extracted into a new flask using a siphon method [29].

The concentration of the soil nanoparticles stocks was obtained at 243 nm by UV-visible spectrophotometry. The stock nanoparticles suspension was diluted to 100 ppm after ultrasonication in column experiments. The pH of the soil nanoparticles suspension and distilled water (DI) used for leaching was adjusted to 4, 6, and 8 using 0.5 M solutions of NaOH or HCl. The ionic strengths (0.1, 0.01, 0.001 M) were regulated by 0.1 M KCl.

2.2. Characterization of soil nanoparticles

The particle size of the soil nanoparticles was measured using an Atomic Force Microscope (AFM) (Multimode-8, Bruker, Billerica, MA, USA) and the size distribution of soil nanoparticle suspension was determined by a Laser Particle Sizer (LS-230, Beckman Coulter, Inc., USA). For AFM, the suspensions of NS were diluted to identical concentrations and applied it evenly on the mica sheet. Particles morphology was scanned using an AFM at room temperature in atmospheric conditions.

The mineral composition of soil nanoparticles was analyzed by X-ray diffraction (D/RAPID II, Rigaku, Japan). The soil nanoparticle suspension was dried to make a powder sample. The sample was spread on a slide and smooth with a glass slide, so that the particles were evenly distributed with a smooth surface.

Organic matter content of soil nanoparticles was measured via Walkley-black titrations [30]. The zeta potential of soil nanoparticles suspension under different IS and pH were determined using a Zeta-Plus (Zetasizer nano ZS90, Malvern Instruments, UK). The Smoluchowski’s formula was used to convert electric mobility of the quartz sand into a zeta potential [31]. The properties of soil nanoparticles were shown in Tables 1–3.

### Table 1. Variation of concentration and physicochemical properties soil nanoparticle suspension with soil type in pH 6, IS = 0.01 M.

| Property                  | AS    | CS    | MS    | RS    |
|---------------------------|-------|-------|-------|-------|
| Concentration (mg L⁻¹)    | 524   | 468   | 358   | 465   |
| Particle size (nm)        | 102   | 97    | 54    | 65    |
| Organic matter (mg g⁻¹)   | 2.3   | 1.0   | 13.5  | 8.2   |

### Table 2. Variation in the particle size of soil nanoparticle suspensions with pH 6, IS changing and IS = 0.01 M, pH changing (unit: nm).

| Factors | AS    | CS    | MS    | RS    |
|---------|-------|-------|-------|-------|
| pH 4    | 110   | 104   | 77    | 80    |
| pH 6    | 102   | 97    | 54    | 65    |
| pH 8    | 90    | 78    | 50    | 57    |
| IS = 0.001 | 95 | 79 | 48 | 63 |
| IS = 0.01 | 109  | 93    | 59    | 68    |
| IS = 0.1  | 213   | 167   | 176   | 149   |

### Table 3. Variation in the zeta potential of soil nanoparticle suspensions with pH 6, IS changing and IS = 0.01 M, pH changing (unit: mv).

| Factors | AS    | CS    | MS    | RS    | Sand |
|---------|-------|-------|-------|-------|------|
| pH 4    | –16.2 | –12.1 | –37.7 | –18.2 | –13.8 |
| pH 6    | –38.1 | –26.3 | –40.6 | –40.9 | –24.7 |
| pH 8    | –43.9 | –18.3 | –35.4 | –47.5 | –29.1 |
| IS = 0.001 | –41.1 | –30.5 | –26.2 | –44.1 | –27.8 |
| IS = 0.01 | –31.2 | –25.5 | –30.9 | –34.2 | –23.3 |
| IS = 0.1  | –26.3 | –20.6 | –21.3 | –23.1 | –15.1 |
2.3. Porous media

Quartz sand (Zhouzhi County Quartz Sand Ltd, China) was sieved to between 0.4–0.5 mm using a stainless steel mesh. Prior to conducting any experiments, the sand was thoroughly cleaned by soaking in a 0.1 M NaOH solution for 12 h, and then with a 0.1 M HCl solution for 12 h to remove any surface metallic ions, and then rinsed with DI water until effluent was clear. The quartz sand was subsequently dried at 105 °C for 24 h.

2.4. Column experiments

NS transport experiments were conducted in columns uniformly packed with quartz sand. The column dimensions were 15 cm length × 3 cm inner diameter. Quartz sand was wet-packaged using deionized (DI) water with stirring to minimize any layering and to remove air entrapment in the column [32]. The porosity of the column was 0.47 and the pore volume (PV) was 50 ml.

The column was initially flushed with DI water for 2 h until the effluent was free of visible impurities. Subsequently, a column experiment was conducted by first injecting 4 pore volumes (PVs) background solutions with different pH (4, 6, 8) and ionic strengths (0.1, 0.01, 0.001 M), respectively. Thereafter the column was leached with 1.2 PVs of soil nanoparticles suspension in the same background electrolyte, followed by flushing with 2 (PVs) of background solution. The pH and IS were kept constant throughout the whole experiment. Column experiments were controlled at a specific velocity of 0.3125 cm min$^{-1}$. Effluent fractions from each column (20 × 10 mL) were continuously collected in test tubes using an automatic fraction collector (BS-16A-LCD, Shanghai Huxi Analysis Instrument Factory Co., Ltd., China). The effluent soil nanoparticles concentrations was analyzed at 243 nm by UV-visible spectrophotometry (UV-2800, Unico, USA) [33]. A tracer element (Cl$^-$, 0.01 M) was used to determine the dispersion coefficient of the hydraulic properties of the columns and the diffusion coefficient could be fitted by software Stanmod. The experimental procedure for the tracer was consistent with the NS. Effluent Cl$^-$ concentration was determined by AgNO$_3$ titration [34]. All column experiments were performed in duplicate.

2.5. Mathematical model

The interaction energy between NSs and quartz sand is determined by the DLVO theory [35]. The total interaction energy ($E_{tot}$) is the sum of van der Waals attraction energy ($E_{vdw}$) and electric double layer repulsion energy ($E_{edl}$):

$$E_{tot} = E_{vdw} + E_{edl}$$

The van der Waals interaction is calculated using the equation [36,37]:

$$E_{vdw} = -\frac{A_r}{6h(1+\frac{146}{h})}$$

where $A_r$ is the Hamaker constant for soil nanoparticles ($1.0 \times 10^{-20}$ J in quartz sand) [38], $r$ is the radius of soil nanoparticles, $h$ is the separation distance between soil nanoparticles and quartz surface, and $\lambda$ is the characteristic wavelength of interaction and was set as 100 nm.

The electrical double layer interaction energy ($E_{edl}$) as a function of separation distance between an infinite plane and a sphere immersed in water is given by equation [39,40]:

$$E_{edl} = \pi \varepsilon_0 \varepsilon_r r^2 \left\{ \frac{2\varepsilon_r \varepsilon_a}{\varepsilon_r^2 + \varepsilon_a^2} \ln \left[ 1 + \exp(-xh) \right] \right\} + \left(\frac{\varepsilon_r^2 + \varepsilon_a^2}{\varepsilon_r^2 + \varepsilon_a^2} \ln[1 - \exp(-2xh)] \right)$$

where $\varepsilon_0$ is the vacuum permittivity (8.85 × 10$^{-12}$ C$^2$/Jm), $\varepsilon_r$ is the relative dielectric permittivity of water (78.5), $\varepsilon_a$ are the surface potentials of soil nanoparticles and quartz sand, respectively, $k$ is the Debye reciprocal length and can be calculated as:

$$k = \sqrt{\frac{2N_A e^2}{\varepsilon_0 \varepsilon_r K T}}$$

where $N_A$ is the Avogadro number ($6.02 \times 10^{23}$ mol$^{-1}$), $e$ is the electron charge ($-1.602 \times 10^{-19}$ C), $I$ is the ionic strength of the background electrolyte, $K$ is Boltzmann constant (1.38 × 10$^{-23}$ J/K), and $T$ is Kelvin temperature (298 K).

The one-dimensional advection-dispersion equation coupled with reaction terms was used to simulate the transport of soil colloids in the water saturated quartz sand column. It was assumed that the interaction between the soil colloids and the sand grains in the column were affected by irreversible-kinetic reactions due to the reversible reaction was almost impossible to observe. The governing equation can be written as [25]:

$$\frac{\partial C_w}{\partial t} = \frac{\partial^2 C_w}{\partial z^2} - v \frac{\partial C_w}{\partial z} - \frac{\rho \partial S}{\theta \partial t}$$

$$\rho \frac{\partial S}{\theta \partial t} = kC_w$$

where $C_w$ is the soil colloid concentration in pore water (mg L$^{-1}$); which reflects the magnitude of equilibrium reaction in the quartz sand column; $D$ is the dispersion coefficient (cm$^2$ min$^{-1}$); $v$ is the flow rate (cm min$^{-1}$); $\rho$ is the medium bulk density (g cm$^{-3}$); $\theta$ is the porosity; and $k$ is first-order kinetic retention constant (min$^{-1}$). In this equation, the release of colloids is neglected, because the release of colloids was shown to too slow during the timeframe of this experiment [41].
As the migration of soil nanoparticles in porous media is still consistent with colloid filtration theory [42], one dimensional advection-dispersion equation was applied to simulate the experimental breakthrough curves, meanwhile the parameter k in the equation can be fitted with the transport fitting software Stanmod [43].

3. Results and discussion

3.1. Effect of NS size

AFM clearly showed the size and shape of the different soil nanoparticles (Figure 1). AS had a relatively larger particle diameter compared with the other soil nanoparticles because the composition of the aeolian sandy soil was mainly quartz, which did not favor the formation of smaller particles [44]. The average particle diameters of NS were 102 ± 8, 97 ± 8, 54 ± 5 and 65 ± 7 nm for AS, CS, MS and RS, respectively.

One limitation of AFM was that was applied only to dry samples. Figure 1 showed that the dispersion of soil nanoparticles was not very homogeneous, it was possible that the soil particles were aggregated during the drying process, which might make the measured average diameter greater than the hydrodynamic size. The size distribution of soil nanoparticles was also determined using a Laser Particle Sizer shown in Figure 2. While most particles were <100 nm, the percentage of particles <100 nm varied with the parent soil, being 68.4, 99.9, 95.3 and 99.7% for AS, RS, CS and MS samples, respectively. The transport of soil colloids in porous medium would be directly affected by the soil nanoparticles size, since larger soil colloids were blocked by porous medium resulting in adsorption and deposition [22]; soil colloidal transport behavior was closely related to particle size distribution [45].

3.2. Electrokinetic properties of NS under different pH and ionic strength

The overall stability of colloids was typically characterized by the suspension zeta potential [46], which was influenced by the suspension pH and ionic strength. Here zeta potential of soil nanoparticle suspension under acidic condition was larger than that of neutral and alkaline condition (Table 3). The zeta potential was previously shown to be affected by the introduction of OH⁻ into the colloidal double layer [47] where higher negative charge resulted in higher repulsive forces which consequently enhanced colloidal stability. The Zeta potential changed significantly with the change of pH, but was far less prominent for MS when compared to other soil nanoparticles. This difference in behavior of the soil nanoparticles may be due to differences in soil nanoparticles mineral composition. Figure 3, XRD pattern of NS, which showed typical spectra of mineral materials.

![Figure 1. The AFM images of different NS: (a) AS, (b) MS, (c) CS, (d) RS.](image-url)
be due to the high levels of organic matter in MS. It had been studied that only when the ionic strength of agricultural soil colloid was less than 0.1 M, the high content of organic matter in agricultural soil colloid would lead to the stability enhanced with increasing ionic strength [50].

### 3.3. DLVO theory

DLVO theory was used to calculate the interaction between soil colloid and quartz sand to predict the behavior of soil colloid transport and retention in porous media [51]. The DLVO interaction energy profiles at the range of pH and ionic strengths were shown in Figure 4. The results showed that the energy barrier increased with the increase of pH value and the decrease of ionic strength. The variation ranges of different NS were 15–220 kT (AS), 10–130 kT (CS), 15–90 kT (MS), 13–150 kT (RS), respectively, indicating that the NS transport readily increased with increasing pH and with decreasing ionic strength. At pH 4.0, the energy barrier between soil nanoparticles and quartz sand is smallest compared to other pH, so the soil nanoparticle is more easily irreversibly deposited to the energy barrier. However, four kinds of NSs showed significant differences at pH 8. AS and RS were difficult to deposit to the energy barrier, which was due to a high energy barrier. MS and CS is easier contrary. And their energy barriers at pH 8 were 220 kT (AS), 67 kT (CS), 98 kT (MS) and 150 kT (RS), respectively. With the decrease of ionic strength, the trend of energy barrier of soil nanoparticles was also the same, but the maximum value of energy barrier was less than pH. The soil nanoparticles were retained in the column under ionic strength occurred in the minimum secondary energy, because they were easily captured at the minimum secondary energy [52]. Figure 4 showed that the 0.1 M ionic strength had a lower primary energy barrier and a deeper secondary energy minimum than other ionic strength. This means that the NS with high ionic strength was easier to deposit in porous media comparison with the lower ionic strength.

### 3.4. Effect of pH on NS transport

Retention of NS in saturated sand column was evaluated through analysis of experimental effluent breakthrough concentration. The breakthrough curves of NS in Figures 5–6 were constructed as plots of the ratio of effluent concentration to the injection concentration (C/C0). The breakthrough curves of NS transport at pH 4, 6 and 8 were shown in Figure 5. The first-order kinetic retention rate constant (k) and the recovery rate were shown in Table 4. The result demonstrated that recovery rate of AS and RS increased with increasing pH. The parameter k decreased with the increase of pH, it confirmed the experimental results of transport of AS and RS. However, the recovery rate of RS was lower than AS.

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**Figure 2.** Particle size volume of NS used a Laser Particle Sizer.

**Figure 3.** XRD spectra of different NS.

Having main and secondary peaks at $2\theta = 30^\circ$ and $8^\circ$, respectively. It could be seen from the XRD pattern that the main mineral composition of NS was quartz and calcite, which also contained a small amount of kaolinite, montmorillonite and muscovite. For the MS, the content of montmorillonite was higher than other NSs. Hence, a change in pH would have a much lesser effect on MS that contained montmorillonite clays because here charge was generated by lattice substitution and was not pH dependent [48].

The zeta potential of nanoparticles generally increased with increasing ionic strength except MS (Table 3) which was attributed to changes in the thickness of the diffusion electric double layer. The diffusion electric double layer of the colloid was known to be thicker at lower ionic strengths, and the repulsion force between the colloid particles was thus larger, which was beneficial for enhanced colloid stability. In agreement with the results observed here the thickness of the soil nanoparticle diffusion electric double layer reduces as ionic strength increased, resulting in a decrease in repulsive force between soil nanoparticles resulting in decreased soil nanoparticle stability and easier flocculation and precipitation [49]. The Zeta potential of MS was the lowest at 0.01 M compared to 0.1 M and 0.001 M. This might
This was attributed to differences in mineral composition of the different soil nanoparticles (Figure 3). The main mineral component of AS was quartz and was similar to the composition of the porous medium used in this study.

Figure 4. DLVO energy between NS and sand under three different pH and ionic strength conditions. The primary energy barriers for NS and sand (a) and the secondary energy minimum for NS and sand (b).
Since calcite that was a metal oxide mineral and kaolinite had a small amount of positive charge compared to others mineral. RS could be more easily studied. Hence, AS were not easily adsorbed to the surface of quartz sand. In contrast, the mineral composition of RS was complex, containing calcite, kaolinite and quartz (Figure 3). Since calcite that was a metal oxide mineral and kaolinite had a small amount of positive charge compared to others mineral. RS could be more easily 

Figure 5. The effect of pH on the transport of NS ((a) AS, (b) RS, (c) CS, (d) MS) in saturated porous media.
The recoveries rate of CS and MS was pH6 > pH4 > pH8, which was different from the change trend of AS and RS. Our experimental results were not consistent with deposited onto the surface of quartz sand. Changes in colloidal solution pH were also known to be influenced by surface properties and chemical composition [1,9].

Figure 6. The effect of ionic strength on the transport of NS ((a) AS, (b) RS, (c) CS, (d) MS) in saturated porous media.
Assume that difference of composition in NS produced the predictions of the DLVO theory. It was reasonable to generally decreased with increasing in $k$ and the $k$ value of the soil nanoparticles initially decreased as IS increased from 0.001 to 0.01 before increasing with increasing ionic strength (Table 5). According to colloidal theory, the diffusion layer of the quartz sand was thicker at lower ionic strengths and the electrolyte ion shielding effect on particle surface charge was weak resulting in stronger mutual repulsion between quartz sand and the soil nanoparticles. When the ionic strength was gradually increased, the surface diffusion layer of the quartz sand and the soil nanoparticles were compressed, and the electrostatic repulsion force reduces, resulting in deposition of particles on the surface of quartz sand [14]. Furthermore, with the increase in concentration of the electrolyte solution, the complexation between the electrolyte and the surface functional groups of the soil nanoparticle were enhanced, and the negative charge on the soil nanoparticle surface would be reduced [49] so that electrostatic repulsion between the particles and the medium was weakened, which resulted in the deposition of soil nanoparticles.

When the ionic strength was 0.001 M, the recovery of AS was greater than that of CS and RS, but was otherwise similar at higher ionic strengths. At the lower ionic strength the differences in the colloidal diffusion layers were mainly due to the differences in the composition of the soil nanoparticles. However, the diameter of soil nanoparticle increased sharply when the ionic strength of the suspension was higher than the critical ionic strength [59], which led to the recovery of the three NSs were low at high ionic strength. With the increase of the ionic strength of the suspension, the double layer thickness of the surface of the soil nanoparticle decreased sharply and the electrostatic repulsion between the particles decreased, and van der Waals force was enhanced by shortening the distance between soil nanoparticle [60]. This promoted the aggregation of particles and the recovery rate was reduced [61].

However, the MS results did not follow this general trend. Compared with the other three soil nanoparticles the changes in recovery and $k$ for MS were different from all other soil nanoparticles, the $k$ value of the soil nanoparticles initially decreased as IS increased from 0.001 to 0.01 before increasing with increasing ionic strength (Table 5). It was also not consistent with the predictions of the DLVO theory. Based on the previous analysis of the Zeta potential of MS, we proposed that MS contained a higher organic matter content and elevated organic contents were low at high ionic strength. With the increase of the ionic strength the differences in the colloidal diffusion layers were mainly due to the differences in the composition of the soil nanoparticles. However, the diameter of soil nanoparticle increased sharply when the ionic strength of the suspension was higher than the critical ionic strength [59], which led to the recovery of the three NSs were low at high ionic strength. With the increase of the ionic strength of the suspension, the double layer thickness of the surface of the soil nanoparticle decreased sharply and the electrostatic repulsion between the particles decreased, and van der Waals force was enhanced by shortening the distance between soil nanoparticle [60]. This promoted the aggregation of particles and the recovery rate was reduced [61].

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4. Conclusion

The behavior of NS transport in the saturated porous media was demonstrated by column experiments and

| Table 4. Best-fit parameter values and recovery rate calculation for soil nanoparticles with varied pH in IS 0.01 mol L$^{-1}$. |
| NS | pH | $k$ (s$^{-1}$) | $R^2$ | Recovery rate (%) |
|---|---|---|---|---|
| AS | 4 | 8.12 × 10$^{-2}$ | 0.96 | 69.75 |
| 6 | 5.51 × 10$^{-2}$ | 0.98 | 79.86 |
| 8 | 2.81 × 10$^{-2}$ | 0.97 | 91.02 |
| RS | 4 | 1.63 × 10$^{-2}$ | 0.94 | 46.45 |
| 6 | 1.09 × 10$^{-2}$ | 0.98 | 70.46 |
| 8 | 8.02 × 10$^{-3}$ | 0.97 | 36.62 |
| CS | 4 | 2.21 × 10$^{-2}$ | 0.97 | 38.62 |
| 6 | 1.40 × 10$^{-2}$ | 0.94 | 53.82 |
| 8 | 2.04 × 10$^{-2}$ | 0.91 | 36.67 |
| MS | 4 | 1.48 × 10$^{-2}$ | 0.84 | 51.99 |
| 6 | 1.04 × 10$^{-2}$ | 0.97 | 62.88 |
| 8 | 1.60 × 10$^{-2}$ | 0.96 | 48.73 |

| Table 5. Best-fit parameter values and recovery rate calculation for soil nanoparticles with varied IS in pH 6. |
| NS | IS (mol L$^{-1}$) | $k$ (s$^{-1}$) | $R^2$ | Recovery rate (%) |
|---|---|---|---|---|
| AS | 0.001 | 2.46 × 10$^{-3}$ | 0.99 | 90.26 |
| 0.01 | 2.60 × 10$^{-2}$ | 0.93 | 29.73 |
| 0.1 | 6.65 × 10$^{-2}$ | 0.45 | 6.05 |
| RS | 0.001 | 1.28 × 10$^{-2}$ | 0.87 | 54.36 |
| 0.01 | 3.18 × 10$^{-2}$ | 0.68 | 24.3 |
| 0.1 | 5.11 × 10$^{-2}$ | 0.78 | 11.29 |
| CS | 0.001 | 1.46 × 10$^{-2}$ | 0.95 | 52.72 |
| 0.01 | 1.63 × 10$^{-2}$ | 0.63 | 18.52 |
| 0.1 | 6.69 × 10$^{-2}$ | 0.54 | 6.08 |
| MS | 0.001 | 1.51 × 10$^{-2}$ | 0.96 | 79.22 |
| 0.01 | 4.76 × 10$^{-3}$ | 0.96 | 79.22 |
| 0.1 | 2.32 × 10$^{-2}$ | 0.53 | 36.24 |
was consistent with the predictions of the DLVO theory. NS transport through quartz sand varied with the origin of the NS and transport decreased in the order AS > RS > MS > CS. NS transport was enhanced with increasing pH and decreasing ionic strength. With some exceptions, attributable to differing chemical composition of the soil nanoparticles, the recoveries of NS generally decreased with increasing ionic strength. It was implied that NS may be released to the groundwater environment during a change in pH and ionic strength of the soil suspension.

Disclosure statement
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