Arsenic oxyanion binding to NOM from dung and aquaculture pond sediments in Bangladesh: importance of site-specific binding constants

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42 sediment

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45 **ABSTRACT**

46 Groundwater contamination by arsenic (As) is a serious public health concern in many
47 different areas worldwide, particularly in the Bengal region. Mobilization and fate of As in
48 natural waters is controlled by a variety of factors including the presence of natural organic
49 matter (NOM). This study experimentally determined conditional distribution coefficients
50 (with K_D defined as the ratio between As bound to NOM and truly dissolved As) and
51 apparent stability constants between As^{III} oxyanions and NOM from cow dung, chicken
52 dung, and Bangladeshi aquaculture pond sediment prior to and after one year of operation.
53 As-sorption experiments with cow dung as the source of NOM resulted in the highest range
54 for $\log K_D$, from 4.7 to 6.3. Pond sediment from Bangladesh after a year of operation for fish
55 production showed greater affinity for binding As oxyanions than fresh sediment prior to
56 fish production. PHREEQCI modeling using constants derived from the experiments along
57 with water chemistry parameters typical for the site supports the dominance of As^{III}
58 oxyanion-NOM complexation in this system. Models employing constants from previous
59 studies using purified NOM considerably underestimate observed complexation by
60 environmental NOM; thus, applying site-specific constants to geochemical models will
61 better predict As speciation for the field site.

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1. Introduction

Arsenic (As) contamination of groundwater is a worldwide problem affecting millions of people. Geochemical modeling is an important and useful tool in predicting how various factors may impact As mobilization and fate in environmental waters. Several studies have shown that natural organic matter (NOM) can play a dominant role in mobilization of As oxyanions through several mechanisms including complexation (Dang et al., 2014; Liu et al., 2013; Wang et al., 2009; Buschmann et al., 2006; Redman et al., 2002); thus, site-specific binding constants for NOM should be included in geochemical models in order to better predict the behavior of As in aquatic systems. Many gaps still remain in our quantitative understanding of the complexation of As with various types of organic matter. Existing models mostly use defined chemistry systems with data from purified NOM and are currently lacking environmental binding constants (Sharma et al., 2010). Further research in both defined and environmental samples is needed.

Arsenic is present in the environment typically as arsenate (As^{V}) or arsenite (As^{III}). In anaerobic conditions, arsenite is more prevalent and exists mostly as uncharged H_3AsO_3 until the first pKa at 9.2. Arsenate exists mostly in an anionic form in environmental waters as the first pKa is 2.2. The second pKa for As^{V} is 6.9 so at circumneutral pH, arsenate will be a mix of H_2AsO_4^- and HAsO_4^{2-} . A mix of oxidation states is often seen, partly due to the presence of microbes that can both reduce and oxidize As. Minerals with solid-phase arsenic can be reduced or oxidized and release dissolved As. Regardless of speciation, both As^{V} and As^{III} are known to sorb to iron minerals including ferrihydrite and goethite.

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87 Research is just beginning to unravel the intricacies of the multiple effects of NOM on As.
88 NOM drives the microbially-mediated reductive dissolution of As-bearing iron (hydr)oxides
89 (McArthur et al., 2001; Harvey et al., 2002). NOM also promotes the microbial processes
90 dissolving minerals for nutrient acquisition which could result in As release (Mailloux et al.,
91 2009). Other interactions between NOM and As include competition for mineral sorption sites,
92 alteration of available surfaces through coating minerals, redox chemistry, and formation of
93 complexes with As oxyanions (Harvey et al., 2006; Bauer and Blodau, 2008; Redman et al.,
94 2002; Davis et al., 2001). Humic acid has been shown to inhibit As^V removal by HFO through
95 competitive adsorption (Du et al., 2014).

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97 NOM can originate from a variety of anthropogenic sources, including agricultural and
98 municipal waste. Several studies and field data from different countries worldwide have shown
99 that As mobilization and the presence of organic matter are related, suggesting that NOM can
100 enhance As solubility through complexation. For example, studies in Taiwan have suggested
101 that As oxyanions bound to humic substance may lead to adverse health conditions, specifically
102 Blackfoot disease (Lu et al., 1990; Wu et al., 1978). NOM is ubiquitous in aquatic systems and
103 contains carboxylic, amino, nitroso, sulfhydryl, hydroxyl, and phenolic groups (Sinha et al.,
104 2011).

105

106 Functional groups in NOM such as carboxyl groups can complex with As (Stevenson, 1992;
107 Mcknight et al., 1992; Wang and Mulligan, 2006; Tessema and Kosmus, 2001; Sharma et al.,
108 2011; Kar et al., 2011; Majumder et al., 2014). Stability constants for binding between As and

NOM from anoxic marine sediments have been reported ranging between 2.0 - 3.2 (in log₁₀) for particulate NOM and between 2.0 - 4.7 (in log₁₀) for dissolved NOM (Dang et al., 2014). Because NOM is a heterogeneous mix of compounds, the goal is not to define exact binding sites or mechanisms; rather, it is important to understand site-by-site variability in NOM characteristics and overall binding affinity.

Additionally, there are very few studies on the role of other ions on the binding between As and NOM, and it appears that other ions can either decrease As binding to NOM through competition or increase it by acting as bridging ions. Lin et al. (2004) suggests that cations, such as Ca²⁺ and Mg²⁺, act in cation bridging in the complexation of As(V) with humic substance. Formation of ternary As(III/V)–Fe(III)–NOM complexes has been identified by spectroscopic evidence in both oxic and anoxic environments (Mikutta and Kretzschmar, 2011; Hoffman et al., 2013). Hoffman et al. (2013) also showed a ligand exchange reaction between As(III) and hydroxylic/phenolic groups in their study of peat reacted with Fe(III) and As under reducing conditions.

One of the areas most severely affected by arsenic mobilization and consequent poisoning is the Bengal region. Concentrations of As in aquifers of Bangladesh and Indian state of West Bengal often greatly exceed the World Health Organization standard of 10 µg/L, and chronic As exposure has resulted in many adverse health outcomes including arsenicosis and cancer, and it is predicted that long-term exposure to current levels of arsenic in water in Bangladesh will result in approximately two million cases of arsenicosis, 125,000 cases of skin cancers and 3,000 deaths from internal cancers per year (Yu et al., 2003).

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133 Recent work has shown that ponds constructed for aquaculture in Bangladesh may be an
134 important source of recharge water containing biologically degradable organic carbon (BDOC)
135 that may mobilize As through reductive dissolution (Neumann et al., 2010, 2011). Additionally,
136 aquaculture practices in the region routinely add agricultural waste, including cow and poultry
137 manure, to the ponds to supply nutrients to stimulate fish production. These practices could
138 potentially have effects on the As mobility by introduction of a new source of NOM.

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140 In this study, we (1) measure and compare conditional distribution coefficients (K_D) and
141 binding stability constants (K_S) for As^{III} oxyanions with four different types of NOM. Two
142 environmental sediment samples were collected from a Bangladeshi aquaculture pond, where the
143 first sample was collected before fish production in the pond and represents the original sediment
144 conditions. To observe effects of aquaculture practices on sediment characteristics, the second
145 environment sample was collected after one year of fish production in the same Bangladeshi
146 aquaculture pond. As the addition of cow and chicken dung is common practice in aquaculture
147 ponds, we also tested NOM extracted from both cow and chicken dung to compare binding
148 affinities of each source. Two different but complementary quantitative parameters, K_D and K_S
149 (which will be introduced later in detail), have been very scarcely reported in the literature for
150 describing the binding characteristics between As and NOM species. These parameters provide
151 complementary information regarding the quantitative capacity of DOM to bind As^{III} and the
152 quantitative stability of the resulting complexes. We also (2) characterize functional groups in
153 the four types of NOM, and (3) test importance of the predicted As^{III} -NOM complexes in
154 PHREEQCI speciation models of pond recharge at a Bangladeshi field site.

2. Methods

2.1 Field Site

Environmental NOM samples were extracted from a constructed pond at the village of Sree Nagar in the Munshiganj district of Bangladesh, approximately 25 km south of Dhaka. Land use in the area is primarily rice and potato cultivation, with rice fields and aquaculture ponds covering the majority of the land area. Collaborators from MIT and BUET excavated and constructed a new pond to serve as an important site for studying the role of ponds in arsenic mobilization.

Immediately after excavation and one year after construction, sediment cores were collected using an AMS Multi-Stage Soil Core Sampler. Cores were capped, sealed, double-bagged in low gas permeable Escal bags containing RP oxygen absorbing agents (Mitsubishi Gas Company), and shipped on ice to UCLA. Cores were collected in six-inch sections reaching a depth of five feet after it was first constructed (prior to a monsoon season) and again after 1 year.

2.2 NOM Extraction

NOM was extracted from four samples (two pond sediment extracts and two dung samples). Sediment from the 2 to 2.5 feet interval in the Bangladeshi pond sediment cores was used. Composite dung samples were collected fresh from chickens and cows located at an animal

sanctuary (Gentle Barn, Santa Clarita, CA). Aliquots of each organic matter source were hand-shaken for 5 minutes with ultrapure Milli-Q water (Millipore) to extract loosely-bound NOM between the range of 90 and 110 mg C/L. The supernatant was filtered (0.45 μ m), deoxygenated, and stored in serum bottles capped with Teflon-coated stoppers at 4°C.

2.3 NOM Characterization

In order to probe the nature of humic substances present, NOM samples were dissolved and separated using ultrafiltration. Then, the dissolved fractions larger and smaller than 5 kDa, referred to as >5 kDa and <5 kDa, respectively, were probed using the polarity rapid assessment method (PRAM), in which samples are run in parallel through solid phase extraction (SPE) cartridges (Rosario-Ortiz et al., 2007; Rosario-Ortiz et al., 2009). C18 was used as a nonpolar SPE representing hydrophobic substances, while diol and amino were used as polar (hydrophilic) and anionic (charged/ion exchange) SPEs, respectively. SPE cartridges contained 100 mg of sorbent and had a total volume of 1.5 mL, with an average pore size of 60 Å. SPE cartridges were wetted with 10 mL of methanol and then rinsed with 200 mL of Milli-Q water. All cartridge filtrates collected during the filter clean-up were analyzed for TOC and UV₂₅₄ to ensure that no cartridge materials were leached out during sample collection. Cartridges were rinsed until TOC measurements were under 0.3 mg/L. NOM water was then pumped through the cartridges using a syringe pump (KD Scientific, Holliston, MA) at a flow of 1.2 mL/min and the filtrate was collected.

In each case, fractions of substances similar to respective SPEs are retained on the cartridges and carbon levels of all fractions are measured via UVA₂₅₄ and/or dissolved organic carbon

(DOC) measurements. DOC analysis was conducted using catalytically-aided Pt 680° C combustion/non dispersive infrared (NDIR) method and a Shimadzu 5050 TOC analyzer. Dissolved NOM was measured as DOC as mg C/L Retention coefficients (RC) were calculated (Equation 1) for each extraction and represent the fraction retained by each sorbent. $C_{\text{breakthrough}}$ is the maximum breakthrough concentration of the sample and C_0 is the initial sample concentration.

$$RC = \frac{1 - C_{\text{breakthrough}}}{C_0} \quad (1)$$

2.4 Nutrient Analysis

Nitrogen and phosphorus levels in cow and chicken dung were quantified through TNTplus (Hach). Nitrogen associated with nitrate and ammonia were measured using dimethylphenol and salicylate spectrophotometric methods respectively, while phosphorus from phosphates was measured using an ascorbic acid method (Hach Company, 2005).

2.5 Dialysis Experiment

Conditional distribution coefficients (K_D) and apparent stability constants (K_S) were determined through a modified method from Buschmann et al. (2006) using dialysis experiments with DOM and As^{III} . 500 Da dialysis tubing (Spectra/Por Biotech Cellulose Ester (CE) membrane) containing 100 mg C/L DOM (size > 500 Da) extracted from one of the four NOM sources was equilibrated inside acid-washed serum bottles filled with a deoxygenate 0-2000 ug/L

As^{III} solution (Fisherbrand sodium arsenite, pH 7, 0.05 M NaCl, 0.15 mM NaN₃, and 1 mM NaHCO₃) and then capped with Teflon-coated stoppers to hold the two ends of the dialysis tubing in place, while maintaining an anaerobic environment. DOM and As^{III}-DOM complexes remain inside the tubing, unable to pass through the 500 Da membrane. Total As inside and outside of the tubing were quantified by Graphite Furnace Atomic Absorbance Spectrometry (GFAAS).

Additional dialysis experiments with cow dung NOM were also conducted to calculate the K_D values at three environmentally relevant levels (10, 50, and 100 mg/L) of calcium (Ca) bracketing natural values in Bangladesh to study the cation bridging effects.

K_D (L/kg DOC) describes the binding between As and NOM and can be used to estimate the amount of bound As based on the mass of DOC ([DOC]) in solution (Equation 2). It must be noted that typically K_D is calculated with the dissolved and bound fractions defined as less than or greater than 0.45 µm, respectively, while here the dissolved fraction is truly dissolved (less than 500 Da). [As]_b is the concentration of bound As and [As]_f is the concentration of unbound or free As in solution. Since the equation for K_D is not derived from reaction stoichiometry between As and NOM, its use is limited to the measurement of conditional K_D estimations.

$$K_D = \frac{[As]_b/[DOC]}{[As]_f} = \frac{[As]_b}{[As]_f \cdot [DOC]} \quad (2)$$

[As]_b and [As]_f represent the NOM-bound and free arsenic concentrations in the system (μg/L) and [DOC] is the concentration of organic carbon in solution (kg/L). Note that the concentration of DOC used here is not the availability of free DOC ligands, as could be assumed from the form of the equation, and thus this K_D is not a typical binding constant, but rather a distribution coefficient (as is the convention for K_D) (Liu et al., 2010; Buschmann et al., 2006).

K_S (M⁻¹) describes the stability of the formed As-NOM complex in system and is calculated based on the stoichiometric relation of As and NOM (Equation 3). [As]_b and [As]_f are the free and bound concentrations of As (M). C_{DOM} represents the molar concentration of effective ligands in DOM (M); however, knowledge of effective ligands in DOM is not necessary for application of a nonlinear model to find the K_S.

$$K_S = \frac{[As]_b}{[As]_f \cdot C_{DOM}} \quad (3)$$

Due to difficulty in describing the molecular structure of DOM, C_{DOM} is often unknown and the following derivation of the As and NOM binding reaction (Equation 4) is used to estimate the K_S in addition to B_{max} (M), the maximum binding capacity of NOM for As (Equation 5).

$$B_{max} = As_b + DOM \quad (5)$$

$$[As]_b = \frac{B_{max} K_S [As]_f}{1 + K_S [As]_f} \quad (6)$$

Fitting Equation 6 to the binding data, parameters K_S and $B_{\max}$ are estimated using nonlinear regression. One simplification is that all binding sites are equivalent.

2.6 Geochemical Modeling

Observed K_S values from the dialysis experiments were used to geochemically model As^{III} speciation in pond water in the presence and absence of DOM using the PHREEQCI geochemical modeling software. Groundwater chemistry was obtained from the Bangladeshi field site by Swartz et al. (2004). NOM-As solution species were added to the PHREEQCI model with binding characteristics specific to each type of NOM studied here. See Supplemental Information for a sample PHREEQCI input and output file.

3. Results

3.1 NOM Characterization

Characterization of chicken dung and cow dung suggests that chicken dung is richer in DOC than cow dung. Although sediment-extracted DOM contains notably less DOC than either types of dung. One-year old sediment contains approximately 10% more DOC than pre-operation sediment (see Table 1). Nutrient analysis showed that cow dung had nearly five times as much P and just over five times as much NO_3 -N as chicken dung, which had 60% more NH_3 -N (see Table 2).

PRAM analysis showed that there are functional differences between cow and chicken dung larger than 5 kDa (see Figure 1). The size fractions below 5 kDa show relatively similar polarity signatures according to RC for each type of SPE cartridge. Dissolved cow dung greater than 5 kDa had six times more polar fractions than dissolved chicken dung. In the same size fraction, dissolved chicken dung contained 1.5 times more anionic fractions and twice as much nonpolar fractions than dissolved cow dung.

3.2 Dialysis Experiment

K_D ranged over two orders of magnitude for the various types of DOM tested here. K_D decreased as the ratio of As^{III} to DOC increases in all cases (see Figure 2), as observed by others (Liu et al., 2010; Buschmann et al. 2006), with K_D decreasing most rapidly for cow dung. K_D for DOM extracted from sediment at a newly excavated pond is low, several times less than that of sediment from the same pond one year later, suggesting that changes that occurred due to operation of pond during that year could have affected the sediment's ability to bind As^{III} . Cow dung showed as much as ten times higher K_D than pond sediment, indicating higher ability to complex As in the aqueous phase (see Table 3). Apparent stability constants for As, or K_S , for cow dung was slightly lower than for the other types of NOM, but because the B_{max} was higher for cow dung, the overall impact is similar sorption.

K_D values for experiments with 50 mg/L Ca and cow manure were comparable with the range of distribution coefficients calculated from the dialysis experiments with pure cow dung. At this level of Ca, higher levels of bound As^{III} resulted in larger K_D values compared to low (10 mg/L)

and high (100 mg/L) Ca concentrations (see Table 4). This might be partially due to competition for binding at higher concentration of Ca. Further studies with varying cation concentrations are needed to investigate the As-cation-NOM bridging in more detail.

3.3 Geochemical Modeling

Arsenic speciation was determined through PHREEQCI using previously characterized groundwater chemistry and the K_S values observed from the dialysis experiments. Results show that in the presence of any NOM, the dominant form of As^{III} is bound to NOM (see Table 5). For a total As concentration of 6.4×10^{-6} M, with no added NOM, the predicted dominant species was H_3AsO_3 at 6.38×10^{-6} M (99.7% of the total), with virtually all of the remaining As as $H_2AsO_3^-$. When each of the four studied types of NOM are added to the model, the NOM-As complex ranges from 73-94% of the total As, with H_3AsO_3 having the second highest concentration.

4. Discussion

NOM can abiotically impact As mobilization through complexation. Results from the dialysis experiments support that the presence of chicken dung and in particular cow dung and Bangladeshi sediment can increase soluble aqueous levels of As, through binding with NOM. Studies have shown increased levels of As in solution in the presence of organic matter such as fulvic acid (FA) and humic acids (HA) (Weng et al., 2009; Sharma et al., 2010).

335 Additionally, both cow and chicken dung have notable fractions of ionically-exchangeable
336 functional groups including carboxylic groups, which has been shown to be important to As
337 release. FA has been indicated to be more effective than HA in releasing As due to its smaller
338 size, higher polarity, and more carboxylic groups (Tessema and Kosmus 2001; Sharma et al.,
339 2010). High K_D 's using cow dung in this study could be partially attributed to the large polar and
340 amino fractions in cow dung. In other work, the extent of complexation varied greatly among
341 NOM samples depending on the origin and cationic metal content and functional characteristics
342 of the NOM (Redman et al., 2002).

344 Aquaculture studies in Bangladesh have been optimizing type, amount, and frequency of
345 addition of manure to deliver the most nutrients with the least risk of anoxia (Hossain et al.,
346 2006; FAO 2001; Yi et al., 2004). However, as the connection between ponds and As
347 contamination was only recently discovered, current aquaculture practice recommendations do
348 not take into account whether conditions could favor As mobilization. It is important to note that
349 some fertilizers, such as cow dung, contain significant amounts of NOM, which can promote As
350 release.

352 The amount and type of NOM in the ponds may be a determinant of downstream As
353 mobilization because NOM plays a key role in As partitioning. Cow dung in particular has low
354 nutrient content and contains tannins that darken the water leading to increased light attenuation
355 and low dissolved oxygen. Recent studies have shown the surface derived DOC from plants and
356 animal products could result in increased mobilized As, including a study by Williams et al.
357 conducted at Bangladeshi rice paddies with high levels of DOC which were fertilized with cow

manure. The study showed considerable increase in mobilized As not attributed to reductive dissolution of iron minerals but perhaps due to interactions with organic matter such as complexation in solution (Williams et al., 2011).

Agricultural fertilization with NOM is also a common remediation technique that contributes to possible additional As-NOM complexation (Bernal 2007, Moreno-Jimenez 2013). The Moreno-Jimenez study has shown that more As is released into pore water where contaminated soil is treated with compost and/or nutrients, implying ionic competition, flooding, resulting pH change, and complexation with As and other bridging ion all impact As release.

5. Conclusions

The outcomes of this research will contribute to a better understanding of interactions between As and NOM by providing the site specific binding constants which can be useful in estimating and modeling As mobilization. Furthermore, recent studies have indicated that environmentally-extracted NOM is more effective than purified NOM at promoting As release, implying that current studies based on purified NOM is likely underestimating effects. Whichever mechanism is at play, the source of organic matter driving the mobilization of As is crucial to understand.

It is also important to further study the role of ponds and the potential impacts of current fertilization practices on As mobilization. Aquaculture practices have increased both the food supply and the standard of living in Bangladesh, and while there is a possibility that aquaculture ponds are linked to increased levels of arsenic in groundwater, the effects of these practices have

not been well studied. Before steps are taken to alter aquaculture practices and/or fertilization remediation techniques, the link between that and As mobilization should be fully elucidated.

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Figure 1. PRAM analysis of cow and chicken dung by UVA₂₅₄.uf denotes NOM in the < 5kDa fraction, while gf denotes NOM in the > 5 kDa fraction.

$$\frac{K_D}{1000} = \frac{81.94}{[As_f]^{0.35}}$$

$$R^2 = 0.83$$

$$\frac{K_D}{1000} = \frac{23.22}{[As_f]^{0.11}}$$

$$R^2 = 0.95$$

$$\frac{K_D}{1000} = \frac{76.52}{[\text{As}_f]^{0.53}}$$

$$R^2 = 0.89$$

$$\frac{K_D}{1000} = \frac{90.84}{[\text{As}_f]^{0.56}}$$

$$R^2 = 0.78$$

Figure 2 (A-D). Conditional distribution coefficients, K_D , as a function of unbound As^{III} for (A) sediment-extracted NOM from a new pond, (B) sediment-extracted NOM from the same pond 1 year later, (C) cow dung-extracted NOM, and (D) chicken dung-extracted NOM. Solid lines are fitted lines based on nonlinear regression.

Figure 3 (A-D). Plots of bound As^{III} as a function of free As^{III} for (A) sediment-extracted NOM from a new pond, (B) sediment-extracted NOM from the same pond 1 year later, (C) cow dung, and (D) chicken dung. Circles represent the experimental data points and dashed lines illustrate the fitted nonlinear binding models.

Table 1. Dissolved organic carbon content normalized to total amount of organic matter (mgC/g)

Organic Matter	DOC/OM
Sediment Year 0	0.89
Sediment Year 1	0.98
Cow Dung	28
Chicken Dung	63

Table 2. Nutrient content of dissolved organic matter from dung (mg/L)

	<i>Nitrate-N</i>	<i>Ammonia-N</i>	<i>Phosphate-P</i>
Cow Dung	0.69	1.5	3.2
Chicken Dung	0.13	2.5	0.66

Table 3. Conditional distribution coefficients (K_D), apparent stability constants (K_S), and maximum binding capabilities of As^{III} to different sources of NOM based on nonlinear regression

Organic Matter	Log K_D (L/kg DOC)	Log K_S (M^{-1})	Log B_{Max} (M)
Sediment Year 0	4.7 - 5.4	5.1	-4.3
Sediment Year 1	4.2 - 4.5	4.9	-4.4
Cow Dung	4.7 - 6.3	4.0	-3.3
Chicken Dung	5.1 - 6.0	5.5	-4.2

Table 4. Conditional distribution coefficients (K_D) of AsIII binding to NOM from cow manure at different environmental levels of Ca.

Calcium Level	Log K_D
Low (10 mg/L)	3.2 - 3.6
Medium (50 mg/L)	4.9 - 5.9
High (100 mg/L)	2.5 - 3.9

Table 5. Predicted speciation of As^{III} (mol/L) in the presence of NOM from difference sources based on PHREEQCI analysis with apparent stability constants determined experimentally in this work. Total $\text{As}^{\text{III}} = 6.4 \times 10^{-6}$ mol/L for all runs. For each column with added NOM, a NOM-As solution species was added to the PHREEQCI model with binding characteristics as given in Table 3. Bangladeshi groundwater chemistry was obtained from Swartz et al. (2004). Modeled concentrations below $1\text{E-}14$ M are listed in the table as Negligible. These modeling results show the dominance of As-NOM species when NOM is included.

As ^{III} Speciation	Ground-water with no NOM	Ground-water with NOM from year 0	Ground-water with NOM from year 1	Groundwater with NOM from cow dung	Groundwater with NOM from chicken dung
H ₃ AsO ₃ -DOM _{sedyr0}		5.50E-6			
H ₃ AsO ₃ -DOM _{sedyr1}			4.67E-6		
H ₃ AsO ₃ -DOM _{cow}				5.39E-6	
H ₃ AsO ₃ -DOM _{chicken}					6.02E-6
H ₃ AsO ₃	6.38E-6	9.18E-7	1.72E-6	1.01E-6	3.78E-7
H ₂ AsO ₃ ⁻	2.5E-8	3.60E-9	6.75E-9	3.97E-9	1.48E-9
H ₄ AsO ₃ ⁺	3.55E-13	5.11E-14	9.57E-14	5.63E-14	2.10E-14
HAsO ₃ ⁻²	2.01E-13	2.98E-14	5.59E-14	3.29E-14	1.23E-14
AsO ₃ ⁻³	Negligible	Negligible	Negligible	Negligible	Negligible