Modeling Nanosilver Transformations in Freshwater Sediments

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Supporting Information

ABSTRACT: Silver nanoparticles (AgNPs), an effective antibacterial agent, are a significant and fast-growing application of nanotechnology in consumer goods. The toxicity of AgNPs released to surface waters during the use or disposal of AgNP-containing products will depend on the chemical transformations the nanoparticles undergo in the environment. We present a simple one-dimensional diagenetic model for predicting AgNP distribution and silver speciation in freshwater sediments. The model is calibrated to data collected from AgNP-dosed large-scale freshwater wetland mesocosms. The model predicts that AgNP sulfidation will retard nanoparticle oxidation and ion release. The resultant AgS-coated AgNPs are expected to persist and accumulate in sediment downstream from sources of AgNPs. Silver speciation and persistence in the sediment depend on the seasonally variable availability of organic carbon and dissolved oxygen. The half-life of typical sulfidized (85% Ag2S) AgNPs may vary from less than 10 years to over a century depending on redox conditions. No significant difference in silver speciation and distribution is observed between ≥50% Ag2S and 100% Ag2S AgNPs. Formation and efflux of toxic silver ion is reduced in eutrophic systems and maximized in oligotrophic systems.

INTRODUCTION

The global market for nanotechnology is estimated to have reached $16 billion in 2010 and is expected to grow to approximately $27 billion by 2015.1 Products containing silver nanoparticles (AgNPs), such as antibacterial cosmetics and textiles, represent a major use of nanotechnology in the consumer goods sector.2 AgNPs are toxic to a wide range of organisms,3 and textiles and cosmetics demonstrate relatively high environmental releases;4 initial estimates suggest AgNP emissions to air, soil, and water during manufacturing, use, disposal, and/or recycling may equal as much as 50% of annual production.5

The toxic effects of as-manufactured, untransformed silver nanoparticles, which are mostly Ag0, have been observed in microorganisms, algae, fungi, vertebrates, invertebrates, and aquatic and terrestrial plants.3 AgNPs tend to oxidize inoxic aquatic environments.7–11 This process releases silver ions (Ag+), which can nonselectively interfere with cell respiration and membrane transport.3 The toxic effects of silver are exacerbated by the tendency of the ionic form to persist and bioaccumulate.12,13

Another particulate species, silver sulfide (Ag2S), forms in the presence of naturally occurring sulfides. In an oxygen-mediated process termed “sulfidation,” the AgNPs react with sulfide to form a surface layer of Ag2S.14 Partial sulfidation results in what has been termed a core–shell structure, although TEM images have shown that complex Ag0–Ag2S morphologies may also occur.3,7,14 Complete sulfidation is possible.14 In 2010, Kim et al. characterized Ag2S NPs found in sewage sludge and proposed that that the sulfide-rich, anoxic environment of sewage treatment plants (STPs) facilitates rapid sulfidation.15 This transformation is of great importance, both because the highly insoluble Ag2S shell reduces the rate of AgNP oxidation7 and because Ag2S is less toxic to microorganisms than Ag0 NPs and Ag+.16–17

The differential toxicity of silver species necessitates the development of environmental fate models that can predict their relative abundance in a system of interest. Although several models have recently been developed to describe nanoparticle transport in environmental media,18–23 no attempts have been made to model the complex chemical transformations of AgNPs in surface waters or sediments.

Smoluchowski coagulation theory indicates nanoparticles will heteroaggregate rapidly with clays, minerals, and other natural colloids upon entering surface water.24 Aggregation is expected to lead to settling, which will lead in turn to accumulation of NPs in sediments.25–26 This conclusion is borne out by laboratory experiments on the behavior of nanoparticles,27,28 as well as the most recent mathematical models of nanoparticle fate and transport in surface waters.18,21–23

The biodiversity and health of aquatic systems can be negatively impacted by the bioavailability of AgNPs and toxic Ag+ by sediment-dwelling organisms. Biomagnification resulting from ingestion of sediment-dwelling organisms by species higher in the food web is a potential concern,29 since trophic transfer of...
NPs has been reported in a simulated terrestrial ecosystem.\textsuperscript{30} Resuspension of particulate silver species and diffusion of silver ion from sediments are also possible sources of silver in the water column.

The model developed in this paper is based on a mass balance model by Di Toro et al.\textsuperscript{31} that describes the speciation of cadmium in sediments in response to redox conditions established in the sediment as a function of oxygen consumption during organic carbon diagenesis, or mineralization.\textsuperscript{32,33} As described below, this adaptation is appropriate because the speciation of both Ag and Cd in sediments is contingent on the distinctions established below. The model was calibrated to experimental data collected from Ag\textsuperscript{0} NP-dosed artificial freshwater wetland mesocosms operated by the Center for Environmental Implications of Nanotechnology (CEINT). The reader is referred to Lowry et al. (2012) for an in-depth treatment of the mesocosm experiment.\textsuperscript{35} After calibration, several input scenarios were selected for further investigation.

\section{Materials and Methods}

\textbf{Model Structure and Framework.} Figure 1 outlines the reactions and physical processes modeled. The diagram distinguishes between oxic and anoxic sediment layers for visual simplicity; however, concentrations of oxygen and all other species were modeled continuously over depth.

Silver ion, Ag\textsuperscript{+}, is derived from the oxidation of the elemental silver (Ag\textsuperscript{0}) core of the AgNPs or from the oxidation of sulfur in silver sulfide. Silver sulfide exists either as a coating on the particle surface, Ag\textsubscript{2}S (NP), or as a free (not bound to the initial AgNP) inorganic precipitate formed by the interaction of iron sulfide with silver ion diffusing freely in the pore water, Ag\textsubscript{2}S (free). This distinction between the two locations of Ag\textsubscript{2}S, “NP” and “free,” was necessary to properly track the extent of silver sorption.

\begin{equation}
2\text{Ag}^0 + \frac{1}{2}\text{O}_2 + 2\text{H}^+ \rightarrow \text{2Ag}^+ + \text{H}_2\text{O}
\end{equation}

Using freely in the pore water, Ag\textsubscript{2}S (free). This distinction was necessary for Ag\textsuperscript{0}, which is assumed to be present only in the AgNP cores. The relevant reactions are as follows:

\begin{equation}
\text{Ag}_2\text{S(NP)} + \frac{1}{2}\text{O}_2 \rightarrow \text{2Ag}^+ + \text{SO}_4^{2-}
\end{equation}

\begin{equation}
\text{Ag}_2\text{S(free)} + \frac{1}{2}\text{O}_2 \rightarrow \text{2Ag}^+ + \text{SO}_4^{2-}
\end{equation}

The main driver of aerobic AgNP oxidation is the depth of oxygen penetration in the sediments, which depends primarily on its consumption during the microbially mediated oxidation of particulate organic carbon (POC):

\begin{equation}
\text{CH}_2\text{O} + \frac{1}{2}\text{O}_2 \rightarrow \text{CO}_2 + \text{H}_2\text{O}
\end{equation}

Silver ion freely diffusing in the pore water is converted to silver sulfide, Ag\textsubscript{2}S (free), via a displacement reaction, in which iron sulfide acts as the source of sulfide:

\begin{equation}
\text{2Ag}^+ + \text{FeS(s)} \rightarrow \text{Ag}_2\text{S(free)} + \text{Fe}^{2+}
\end{equation}

Iron sulfide serves as a proxy in the model for all available sulfides in the system (e.g., acid-volatile sulfide, or AVS). Iron sulfide is gained via the anaerobic oxidation of organic carbon, which reduces sulfate to sulfide,

\begin{equation}
\text{2CH}_2\text{O} + \text{SO}_4^{2-} \rightarrow \text{S}^{2-} + \text{2CO}_2 + \text{2H}_2\text{O}
\end{equation}

and is lost by oxidation to form iron oxyhydroxide (FeOOH):

\begin{equation}
\text{FeS(s)} + \frac{9}{4}\text{O}_2 + \frac{3}{2}\text{H}_2\text{O} \rightarrow \text{FeOOH(s)} + \text{2H}^+ + \text{SO}_4^{2-}
\end{equation}

Finally, iron oxyhydroxide and particulate organic carbon reversibly sorb silver ion.

\begin{equation}
\text{Ag}^+ \rightarrow \text{Ag} \equiv \text{POC}
\end{equation}

\begin{equation}
\text{Ag}^+ \rightarrow \text{Ag} \equiv \text{FeOOH}
\end{equation}

Physical processes modeled include particle mixing due to bioturbation, diffusive mixing of dissolved species, efflux of dissolved species as a result of diffusion at the sediment-water interface, and influx of organic carbon, AgNPs, and oxygen. Note that, in the tradition of conventional mass balance models, “particle” refers here to all particulate (solid phase) species: POC, FeS, FeOOH, Ag\textsuperscript{+}, Ag\textsubscript{2}S (“NP” and “free”), Ag\equiv POC, and Ag\equiv FeOOH. “Nanoparticle” refers only to the particulate
species representing the core and shell of the transforming AgNP (Ag₀ and Ag₂S (NP), respectively).

Supporting Information (SI) Table S1 lists the model state variables. Changes in the depth profile of each state variable over time are modeled dynamically using the one-dimensional continuous advective-dispersive mass balance equation:

$$\frac{\partial [C(z)]}{\partial t} - \frac{\partial}{\partial z} \left( \left( f_p D_p(z) + f_d D_d \right) \frac{\partial [C(z)]}{\partial z} \right) = \sum_{j} R_j$$  \hspace{1cm} (11)

where $C(z)$ is the concentration at depth $z$, $f_p$ is the particulate fraction of the total concentration, $D_p$ is the particle mixing coefficient, $f_d$ is the dissolved fraction of the total concentration, $D_d$ is the diffusive mixing coefficient, and $\sum R_j$ represents all reactions resulting in gain or loss of the species.

Reactions. In general, reaction rates were assumed to exhibit a linear dependence on the concentrations of all reactants.\(^3\) The exceptions to the assumption of linear dependence were aerobic and anaerobic oxidation of organic carbon, for which we applied the Michaelis Menton expression for oxygen dependency. SI Table S2 summarizes the reaction equations, which are of the general form

$$R_j = k_i \theta_j (T-20) [C_1][C_2]$$  \hspace{1cm} (12)

where $k_i$ is the reaction rate constant, $\theta_j$ is the Arrhenius temperature coefficient, $T$ is the temperature in degrees Celsius, and $[C_1]$ and $[C_2]$ are the concentrations of the reactants.

Previous work suggests the rate constants for AgNP oxidation and AgNP sulfidation, $k_{AgO}$ and $k_{Sulf}$, decrease exponentially in response to AgNP sulfidation.\(^7,14,36\) The AgNP oxidation rate was thus modeled using eq 13, where $S/Ag$ represents the molar ratio of sulfur to silver. AgNP sulfidation, in contrast, could not be accurately modeled using eq 13 without drastically reducing the time step (and vastly increasing model runtime). For fully unsulfided AgNPs, the rate of sulfidation was assumed to decrease rapidly upon dosing so that a constant “long-term average” sulfidation rate would be appropriate over the time scale considered (see SI Table S3 for details). The effect of additional factors (e.g., temperature, pH, organic carbon, particle size, capping agent)\(^8,10\) on the initial rate of AgNP oxidation before sulfidation ($S/Ag = 0$) was captured by calibration.
**Partitioning.** For simplicity, all dissolved sulfides were modeled as the dissolved form of FeS using the partitioning coefficient \( \psi_{FeS} \). The governing equations for iron sulfide “partitioning” are provided by Di Toro et al. (1996). Equilibrium partitioning of silver between the sediment pore water (Ag\(^+\)) and the relevant solid phases (Ag\(\equiv\)POC and Ag\(\equiv\)FeOOH) was described with a Langmuir isotherm analogous to that described by Di Toro et al. for cadmium. \(^{35}\) Details are provided as SI (equations S1–S7). The complexion of Ag with chloride ion, while strong, is significantly less so than for sulfide and for organic matter, \(^{37}\) and is assumed to be negligible in the freshwater (low chloride) environment. AgCl(s) was not detected by XAS measurements on the mesocosms sediments. \(^{35}\)

**Particle Mixing and Porosity.** The particle mixing coefficient, which represents the rate of sediment mixing as a result of biological activity, is temperature-dependent and is assumed to decrease exponentially with depth, \( z \), according to the equation:

\[
D_\phi(z) = D_\phi(20) e^{-z/z_p}
\]

where \( z_p \) represents the depth of bioturbation.

Since sediments at depth are compacted by the weight of overlying layers, porosity was assumed to decrease exponentially with a decay constant of \( k^p \) from a value of 0.99 at the interface \( \phi(0) \) to a value of 0.25 at depth. Porosity at the interface and at depth was estimated from mesocosm data \(^{35}\) which produced good agreement between silver concentrations vs depth observed in the mesocosms and those predicted by the model.

**Temperature.** Annual temperature variation is assumed to be sinusoidal. Coefficients were estimated from water column temperatures in the mesocosms used to calibrate the model \(^{35}\) and air temperature data at the mesocosm site (http://ceint.duke.edu/chart/mesocosm-air-temp). Higher temperatures increase rates of reaction and solute diffusion via Arrhenius temperature coefficients (not shown in Table 1; see SI Table S3) and correspond with increases in biological activity in the sediment and water column. This leads to higher particle mixing, higher organic carbon influx to the sediment, and the depletion of dissolved oxygen.

**Parameter Values.** Table 1 summarizes parameter values used in the simulations. SI Table S3 presents the assumptions underlying the selected values. Sensitivity analysis revealed that model outputs were not overly sensitive to local variation in those parameters for which only point estimates were found (SI Table S4).

The nominal values of four model parameters (\( k^p, D_\phi, K_{\infty}, k^{Ag,0}_{Ag,0}(S/Ag=0) \)) were determined by calibration. System parameters were matched to conditions in the mesocosms, including loss on ignition (L.O.I. \( \approx 2k_{\infty} \)), \(^{36}\) where \( k_{\infty} \) is the initial organic carbon composition of the sediment), sediment AVS, oxygen concentration at the sediment-water interface, annual temperature variation, AgNP mass input, and time of dosing. \(^{35}\) The simulation was run for \( t = 18 \) months after introducing a simulated pulse input of 2.9 g of “pristine” (unsulfidized) Ag\(^+\) NPs. The AgNPs added to the mesocosms had a thin oxidized Ag (e.g., Ag\(_2\)O) shell, \(^{36}\) which was an implicit determinant of the initial rate of nanoparticle oxidation, \( k_{n}^Ag^0(\text{Ag}^{(s)}=0) \). For the calibration, mean total silver concentrations calculated by the model over sediment depths of \( 0–1 \) cm, \( 1–2 \) cm, \( 2–4 \) cm, and \( >4 \) cm were matched to the median silver concentrations observed in the mesocosms eighteen months after dosing. Model outputs were also matched to the relative abundances of Ag\(^0\), Ag\(^+\)-organics, and Ag\(_2\)S in the surficial sediments of the mesocosms eighteen months after dosing, as determined from linear combination fits of X-ray absorption spectroscopy (XAS) spectra collected on surficial sediment in the mesocosms. \(^{35}\) The calibration was performed using the nonlinear parameter estimation software PEST (http://www.pesthomepage.org/), which minimizes the weighted sum of the squared residuals. Weights were calculated as the inverse standard deviation of each field observation.

Silver ion efflux from the sediment to the overlying water at time \( t \) was estimated by the model from the predicted concentration profile of Ag\(^+\) at the sediment-water interface.

**Solution Method.** A fully implicit finite difference approximation was used to simultaneously solve the mass balance equations for all state variables. The Gauss-Seidel iteration method (e.g., as described by Ramaswami et al. \(^{39}\)) was used to achieve rapid convergence. To avoid dynamic instability associated with the central differencing approach to numerical methods, \(^{39}\) model resolution \( (\Delta z = 1 \text{ mm}, \Delta t = 0.05 \text{ days}) \) was chosen to ensure

\[
\Delta t \leq \frac{(\Delta z)^2}{2D_{max}}
\]

where \( D_{max} \) was the maximum value of the time- and depth-dependent diffusion coefficients \( D_{\phi} \) and \( D_{\phi} \).

An exponentially expanding spatial grid was implemented in order to focus computational efforts on transformations occurring near the sediment-water interface while eliminating undue computational burdens at depth (see SI equations S8–S17 for details). \(^{46,47}\) Model results for the expanded grid were compared to those for an unexpanded grid to affirm the accuracy of the method.

The diagenetic model was run for a year before simulated introduction of the AgNPs to ensure periodicity in seasonally variable redox conditions had been achieved.

**Initial and Boundary Conditions.** No-flux boundary conditions were assumed at the sediment-water interface for all particulate species except those entering the system—POC, Ag\(^0\), and Ag\(_2\)S (NP)—which assumed known influx (Type II/Neumann) boundaries. This boundary condition was assumed to be sufficient to capture the net effect of particle deposition and resuspension in a quiescent wetland environment. Influx of organic carbon was described piecewise in time with a sine curve that peaked at \( f_{POC,max} \) in summer (mid-May to mid-September) and was 15 mg/m\(^2\)-d in fall, spring, and winter. This idealized influx scenario agrees with seasonal trends in community respiration and carbon mineralization observed in a freshwater marsh. \(^{48}\) Known constant concentration (Type I/Dirichlet) boundaries were assumed for all dissolved species. ICP-MS analysis of mesocosm water column samples detected no total silver after initial settling (detection limit of \( <2 \) µg/L), \(^{35}\) so it was assumed that the silver ion concentration at the sediment–water interface could be approximated as zero. At \( t = 0 \), the oxygen concentration was decreased linearly from its value at the interface to zero at a depth of 5 mm. The depth profile of oxygen
exhibited exponential decay within one day of launching the simulation.

The mesocosms were dosed with 4.2 g of polydisperse, partially oxidized (80–85 wt % AgO) 30–80 nm PVP-coated AgNPs, resulting in initial water column concentrations of 25 mg/L AgNPs. Ag0 was assumed to exhibit an exponentially declining flux such that all silver added to the water column as a finite square pulse input would penetrate the sediment within 10 days, as observed in the mesocosms. At depth, no-flux boundary conditions were implemented for all species.

RESULTS

Figure 2 compares total Ag concentration vs depth as measured in the mesocosm sediment to the simulated concentration profile.

Figure 2. Total sediment silver concentration profile predicted by the model (curve) versus sample data collected from the water-column-dosed mesocosms (38–80 mm depth) at each of four sediment depths (0–10 mm, 10–20 mm, 20–40 mm, >40 mm). Crosses (×) indicate mean values.


de
df

The calibrated value of the silver ion-organic carbon partitioning coefficient is on the upper end of its expected range (log Koc = 4.1–7.8 L/kg). This suggests the organic carbon in the mesocosms sediments had a relatively high affinity for Ag+, perhaps due to the presence of strong chelating groups like reduced sulfur. The overall correlation coefficient representing agreement between model output and mesocosm data is 0.997.

Figure 3a shows the silver ion efflux from the sediment to the overlying water as a function of time for a simulated pulse input of 2.9 g of AgNP under typical environmental conditions for a freshwater wetland (the "nominal" case; see bolded values in Table 1). The modeled system was "dosed" in July in order to maximize initial efflux. This represents a worst-case scenario, in which the maximum amount of toxic Ag+ escapes into the water column before the nanoparticles have the chance to sulfide and AgNP oxidation rates decrease.

Changes in the depth profile of all state variables over time are provided in SI Figure S2. Oxygen concentration peaks in winter, as has been observed in natural systems. However, Ag+ efflux peaks in summer. SI Figure S3 reveals the mid-July peak in Ag+ efflux is driven by the temperature dependence of mixing and reaction rates.

The five scenarios in Figure 3 demonstrate the role sulfidation plays in the release of Ag+ from AgNPs. Scenarios include an unrealistic worst case, in which a pulse input of pristine Ag0 NPs enters the system (t = 0) and the AgNP sulfidation process is artificially turned off (i.e., nanoparticles remain 100% Ag0 over the course of the simulation). The other four scenarios demonstrate system responses to a dose of pristine particles (0% AgS before dosing), fully sulfidized particles (100% AgS before dosing), and partially sulfidized particles (50% and 85% AgS before dosing) when sulfidation is allowed to occur. This last scenario (85% AgS) reflects the extent of sulfidation expected for silver particles exiting a sewage treatment plant by way of effluent. Figure 3b shows depth profiles under these four scenarios one year after dosing.

We observe no differences in the extent of AgNP oxidation and the speciation, solid phase partitioning, and efflux of silver ion over time between 50%, 85%, and 100% AgS nanoparticles. This is because the rate of AgNP oxidation decreases exponentially with the extent of AgNP sulfidation (eq 13). AgNPs that enter the system fully sulfidized similarly behave as if fully sulfidized within a year of dosing, although elevated Ag+S (free) concentrations are observed as a result of AgNP oxidation—and subsequent reaction of the released Ag+ with sulfide—within the first several months.

Figure 3b (top right panel) reveals that the predominant form of silver in the sediment will be the partially or fully sulfidized nanoparticles. Percent sulfidation decreases with depth in this pulse input case because the AgNPs that remain at the interface longer spend more time in the presence of O2. Although FeS and O2 do not coexist at equilibrium, O2 mediates AgNP sulfidation. The principal forms resulting from speciation and solid phase partitioning of Ag+ released during AgNP oxidation are the thermodynamically favored species, Ag2S (free), and Ag≡POC (silver associated with particulate organic carbon). This is unsurprising, since silver reacts strongly with the sulfur-containing functional groups prevalent in organic compounds. Ag≡POC may be especially high in this model because of the high Koc value determined by calibration. Truly dissolved (highly toxic) Ag+ is present only at very low concentrations, consistent with observations in field samples.
The normalized sensitivities of model outputs to small (±0.05%) changes in model inputs around the nominal case (i.e., elasticities), reported in SI Table S4, reveal that environmental conditions strongly affect the system response. Figure 4 describes oxidation, sulfdation, and persistence of the AgNPs as a function of organic carbon content.
Aerobic diagenesis of carbon in high organic carbon systems results in low oxygen availability, and subsequent sulfate reduction results in high sulfide availability. Thus Ag⁺ efflux occurs less readily (Figure 4, top left panel) but sulfidation occurs more readily (Figure 4, bottom left panel) in high OC systems. The half-life of the 85% sulfidized AgNPs was found to vary from 6.6 years to as much as 280 years in the model depending on oxygen availability (Figure 4, right panel). AgNP half-lives were relatively insensitive to whether the particles were partially sulfidized or pristine (100% Ag₀) upon dosing—half-lives for the pristine AgNPs ranged from 3 months (low OC) to 280 years (high OC). Loss of Ag₀ from the AgNP cores (SI Figure S4) results from Ag⁺ formation and sulfidation, and does not exhibit first-order exponential decay.

The mesocosm experiment used a single pulse input of a relatively high concentration of AgNPs. A continuous input of low concentrations of nanoparticles from a wastewater treatment plant is a more realistic scenario. We assumed a continuous input scenario in which 20 million gallons per day of STP effluent (76,000 m³/d) containing 33 μg/L 85% sulfidized AgNPs were released to a small (75 km²) lake, and that all silver entered the sediments. Results for the constant input case are provided in SI Figures S5 and S6. Trends agree with those observed in the pulse input case (Figure 4). AgNP accumulation occurs over time in this chronic low dose input case.

**DISCUSSION**

Nanoparticles are expected to accumulate in sediments after release to surface waters. The toxicity of surface-reactive nanoparticles in environmental media depends greatly on chemical transformations undergone by the nanoparticles and released ions. Recent work has shown particle properties strongly affect AgNP transformations.6–11 The present work reveals environmental conditions also play an important role.

Particulate organic carbon (POC) diagenesis has long been recognized as a key determinant of oxygen penetration depth and a driving force for sediment processes. The model suggests this occurs because the pristine AgNPs ranged from 3 months (low OC) to 280 years (high OC). The model with no explicit inclusion of aggregation can successfully reproduce observations from a freshwater mesocosm study.35 In several environmental risk studies of nanosilver,9–21 predicted environmental concentrations (PECs) were compared to no observed effect concentrations (NOECs) or LC₅₀ values for the pristine Ag₂NPs. Nanosilver risk is overestimated by this approach, which overlooks the significant reduction in acute toxicity expected after AgNP sulfidation.9 To improve risk estimates, sediment PECs should be compared to toxicity thresholds for partially and fully sulfidized AgNPs.

Even under oxic conditions, the toxic species Ag⁺ represents <0.01 wt-% of the total silver in the system. Thus the toxicity of environmentally transformed AgNPs and released ions may be quite low. However, AgNP risk may be underestimated if, as has historically been the case, Ag⁺ is considered the only bioavailable silver species in freshwater environments.13,29,34 Ingestion, uptake, or surface interactions with unsulfidized or partially sulfidized AgNPs by aquatic plants and sediment organisms may lead to higher exposures.35

Because Ag₂S is the principle product of AgNP transformation, the rate of oxidation of sulfur in Ag₂S is of particular interest. Previous work reveals that complexation with metal ions stabilizes thiols and sulfides against oxidation under aerobic conditions.5 Although Ag₂S is highly stable against dissolution (K_d = 5.92 × 10⁻¹³),5,53 Ag₂S can exhibit greater solubility in the presence of sediments. The authors suggest this occurs because Fe(III) in the sediments facilitates oxidation.42

At concentrations orders of magnitude higher than observed here (≥10 mmol Ag/m³), Ag⁺ has been shown to transform into Ag₂NPs in the presence of reduced humic acids.5,55 This reaction occurs most rapidly in anoxic and suboxic environments. Because FeS and organic matter act as competing ligands in such environments, we do not anticipate that the in situ formation of Ag₂NPs will play a significant role in nanoparticle fate for lower releases of Ag⁺ in sediments.

Future experimental work may reveal that other oxidants (e.g., Fe³⁺, Mn⁴⁺, NO₃⁻, SO₄²⁻) can oxidize AgNP in sediments under nonequilibrium conditions. One strength of our approach is the ease with which different oxidants or transformative processes (e.g., bacterially mediated oxidation of sulfur in Ag₂S) can be incorporated into the model.

Regulatory and industry decision-making for the safe production, use, and disposal of nanomaterials requires models that can track the environmentally relevant species. Although process-based models capable of describing the temporal evolution of particle size and aggregation state are highly desirable, they present unique theoretical challenges and computational demands. This work reveals that a mass balance model with no explicit inclusion of aggregation can successfully reproduce observations from a freshwater mesocosm experiment.

By omitting aggregation, the model assumes AgNPs mix in the sediment at approximately the same rate as the sediment particles themselves. Since AgNPs are largely expected to heterogeneous (and thus cotransport) with sediments in natural environments, this assumption appears justified. However, since the overall particle mixing rate (D_p) was determined by calibration,
there is no way to test the assumption in this work. In general, data collected at high spatial and temporal density are needed on another AgNP type or on a different environmental system to validate the model. Nonetheless, we believe the incorporation of nanoparticle chemistry into a conventional mass balance sediment metal modeling framework represents a significant step toward more accurate environmental risk models for nanoparticles.56

ASSOCIATED CONTENT

Supporting Information
Supporting Information provides model details, including reaction equations and implementation specifics for an exponentially expanding finite difference grid. SI also includes additional model output for pulse inputs and constant inputs of AgNPs. This material is available free of charge via the Internet at http://pubs.acs.org.

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Notes
The authors declare no competing financial interest.

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