SECTION 1: INTRODUCTION AND BACKGROUND

Since the advent of the nuclear age, the fate of anthropogenic and naturally generated radioactive isotopes in the environment has been a major focus of groundwater transport studies. These radioactive isotopes can present a significant human and environmental health hazard, but also have already provided unique tracers for a variety environmental processes. For example, vertical groundwater movement has been traced using radioactive isotopes generated from atmospheric bomb testing (Solomon, Poreda, Schiff, & Cherry, 1992) and iodine isotope changes in local environments have been correlated with nuclear accidents such as the one at Fukushima (Landis et al., 2012). Radium was used as a phosphorescent compound historically, but sees little, if any industrial use today, instead posing a natural radiological hazard. As an example, radium at secular equilibrium can be liberated from deep formation waters by anthropogenic or natural processes, posing both a direct and indirect hazard. Hydraulic fracturing, in particular, has recently raised some concerns due to the elevated presence of radium in produced waters, with concentrations in excess of 120 Bq/L (Barbot, Vidic, Gregory, & Vidic, 2013). In other situations, radium poses a risk as the source of cancer causing radon, which can accumulate in poorly ventilated basements (Nazaroff, 1992). Experimental work studying radium transport can be used to build groundwater transport models that then inform policy decisions surrounding these scenarios.

In parallel, the radium isotopes also have been leveraged as naturally occurring environmental tracers. The primary example uses a mass balance of radium isotopes in estuarine and near shore systems to provide estimates of subterranean groundwater discharge (SGD). The method constructs a simple conservative mixing model of local groundwater containing naturally occurring radium isotopes to reach a desired off shore end member (Moore, 2003). While many of the predictions of this model have matched well with other methods, it is not able to resolve groundwater behavior within the estuarine systems, especially given that it is well understood radium isotopes are retarded by the presence of iron and manganese oxides (Gonneea, Morris, Dulaiova, & Charette, 2008). This is complicated by the highly transient nature of these systems, which experience rapid changes in solution salinity, pH, and redox state. Aside from a natural groundwater tracer, radium has also been identified as a possible marker for historic spills associated with hydrocarbon extraction (Lauer & Vengosh, 2016). This is possible because the radium isotopic signature of many shales differs significantly from that of local shallow groundwater (Warner, Christie, Jackson, & Vengosh, 2013). Making full use radium as a tracer requires, however, a careful accounting of the critical processes controlling transport.

Radium isotope transport is dominated by advection and diffusion, where specific mineral phases control the aqueous radium concentration through sorption. Radium weakly forms complexes with carbonate and sulfate ions at pH values greater than 9, but is primarily Ra+2 for most environmental conditions (Grivé, Duro, Colàs, & Giffaut, 2015). Previous work has studied sorption by measuring and comparing distribution coefficients, Kd, finding that iron and manganese oxides dominate radium retention (Bassot, Stammose, & Benitah, 2005; Beneš, Strejc, Lukavec, & Borovec, 1984). While a useful indicator for the extent of radium retention in a given system, these coefficients are primarily empirical and are not easily adapted to dynamic systems or systems with differing mineral composition. As an example, a compilation of radium isotope Kd values saw significant variations in Kd for similar systems, even when only a synthetic iron oxide was considered (Beck & Cochran, 2013). More recent work has begun to address this by modeling data from radium sorption to ferrihydrite and goethite using a tetradentate binding site surface complexation model (Sajih et al., 2014).

The objective of this work is to develop further understanding of radium sorption behavior and develop simple models of radium sorption that can be used in a wide range of groundwater studies. Previous work has almost entirely focused on the importance of iron oxides, but have not considered the role of clay minerals with exchangeable cations, or the importance of reduced iron minerals, both of which are freely available in many groundwater scenarios. In this study, we first compare sorption of radium to ferrihydrite, goethite, sodium montmorillonite, and pyrite with a low salinity background solution, and then model that sorption through simple surface complexation models. The results show that pH plays a crucial role in determining sorption extent, and that while iron oxides do show extensive sorption, the most dominant mineral is in fact sodium montmorillonite. [CONCLUDING STATEMENT PERHAPS RELATING TO THE SURFACE COMPLEXATION MODEL RESULTS. NORMALIZED BEHAVIOR]

SECTION 2: METHODS

Reagents used in the experiments were of reagent grade or better, and all solutions were made with 18 M\Ohm water. Radium-226 stock was provided by the MIT Environmental, Health, and Safety office, and acidified to 3% using HCl.

2.1 MINERAL PREPARATION

Both ferrihydrite and goethite minerals were prepared using standard methods (Schwertmann & Cornell, 2000). Briefly, ferrihydrite was precipitated by dissolving Fe(III)Cl3 in water, and rapidly titrating the solution to pH 7, followed by centrifugation and washing to remove background electrolytes. The prepared ferrhydrite slurry was characterized using the ferrozine method (Stookey, 1970), and aliquots of the slurry were added directly to the experiments. Goethite was prepared by slow oxidation of an O2 free solution of Fe(II)Cl2 and bicarbonate using air over the course of 2 days. The resulting mineral was centrifuged and washed to remove background electrolyte, and then dried for 2 hours at 70 C. Both iron minerals were characterized using x-ray diffraction to confirm their composition.

Calcium montmorillonite STX-1b was ordered from the clay minerals society, and was fractionated and washed using standard methods (THAT BOOK). The clay was dispersed with 1 M NaCl, and the <0.2 um clay fraction siphoned off after successive washes with DI water. The siphoned clay particles were then treated with a 1 M Sodium Acetate solution, set to pH 5 using glacial acetic acid, which removed the carbonates. Since the mineral data from the clay society indicated low or negligible iron content, no iron oxide removal was performed. The cleaned clay was then washed and centrifuged with the experimental background solution, resulting in a sodium montmorillonite. The clay was dried at 50 C overnight, and then carefully ground using mortar and pestle. (PROBABLY NEED TO CHARACTERIZE THAT CLAY)

Lastly, pyrite was ordered from (X COMPANY), which came in a large ore form. The pyrite was was ground using mortar and pestle, and then the 45-250 um size fraction were sieved out. The pyrite was placed into an anaerobic glove bag, with a 5% hydrogen and 95% hydrogen atmosphere. Once in the glove bag, the pyrite was washed in 6 N HCl overnight to remove any iron oxide coatings, and then washed with DI water. Lastly, the pyrite was allowed to air dry, with a dessicant nearby. [CHARACTERIZATION WITH SOMETHING].

2.2 SORPTION EXPERIMENTAL SETUP

200 mL serum vials were filled with 100 mL of 10 mM NaCl stock solution, 30 mg of one mineral (except for the case of pyrite, where 40 mg was used), and 5-270 Bq of Radium-226 Stock. Experiments using pyrite were performed in an anaerobic glove bag with oxygen free solutions, while all others were performed in lab air. The pH was titrated to 3,5,7 or 9 +/- 0.05, and then the bottle was sealed with a butyl stopper. Bottles were shaken for 24 hours, and then the pH was checked and re-titrated to the desired value. Previous work has established that this is sufficient time for sorption equilibrium to iron surfaces (Sajih et al., 2014), while sorption to montmorillonite was evaluated using the same set up with different time points. If the pH deviated more than 0.1 pH units, then the bottle was allowed to equilibrate for 15 minutes, and the re-titration process repeated. HCl and NaOH at high concentrations were used, so that volume additions did not exceed 5% of the original volume. Once re-titration was complete, the samples were filtered using a 0.2 um polyethersulfone filter, which was shown not to significantly sorb radium. Solids for ferrihydrite and goethite were collected on the filters for further analysis and to check mass balance. Error in these measurements was quantified by performing points in triplicate and measuring the standard deviation of those results.

2.3 ANALYTICAL TECHNIQUES

Solutions of radium were quantified using scintillation counting techniques. 10 mL of sample (5 mL of Sodium montmorillonite supernatant due to filtration difficulty) were mixed with 10 mL of Ultima Gold XR (Perkin Elmer) and sealed for 30 days to allow radium-226 to reach a transient equilibrium with its daughter products. The equilibrated samples were then counted using a Beckman Coulter scintillation counter (COUNTER INFO), and the resulting counts were compared to a calibration curve of radium-226 standards to determine solution activities. Except for points involving ferrihydrite at pH 9, this was sufficient to determine the extent of sorption and develop isotherms. Background concentrations were also quantified to develop a limit of blank of 1.4 counts per second (cps). We only considered samples having 1.5 times that limit in subsequent analyses.

Many supernatant samples using ferrihydrite at pH 9 were below this defined detection limit, so solid associated radium on the ferrihydrite itself was counted using gamma spectroscopy in addition to the scintillation counted supernatant. (DETAILS ON THE GAMMA COUNTER) was calibrated using a multinuclide standard prepared from (???). The solid samples on PES filters were simply placed in scintillation vials, and counted directly on the counter, with the resulting counts being adjusted for ferrihydrite loss during filtration.

[DISCUSSION OF SURFACE AREA ANALYSIS]

2.4 SURFACE COMPLEXATION MODELING

Surface complexation modeling used a double diffuse layer (DDL) model to account for electrostatic interactions. Simple single site models were used to fit the data, which allows for easy comparison of the relative importance of the different minerals for radium retention, as well as comparison with currently existing surface complexation modeling results (Dzombak & Morel, 1990). In the case of sodium montmorillonite which has been shown to exchange cations, we also posed an exchange reaction following previous surface complexation models of clays (Bradbury & Baeyens, 2005).The relevant reactions are shown in table 1. Experimental sorption data was fitted to PHREEQC model runs until a good visual fit was achieved by varying only the reaction constants for radium complexation with the surface and exchange with clay inner layers. Site densities were set using the BET surface area measured previously, and exchangeable sites determined by the cation exchange capacity reported by Clay Society. Solution complexation behavior was accounted for using the SIT database, which covers radium carbonate, sulfate, and hydroxide complexes, and modeling results showed that carbonates would play a minimal role in high pH experiments.

SECTION 3: RESULTS AND MODELING

SECTION 3.1 SORPTION ISOTHERM RESULTS

Figure 1 shows a set of four isotherms at pH 7, comparing sorption of the four minerals studied. Lines were fit to each set of mineral data, showing R2 values of 0.98 and greater, indicating very linear response for this range of radium activities. A complete data set, showing each mineral, and the associated sorption at different pH values can be seen in the supporting information. The circumneutral pH highlights trends seen at the other pH values. Sodium montmorillonite sorbs radium extensively at this pH, with 85+/-1% of the total radium associated with the solid. In contrast, ferrihydrite only sorbed 50+/-7% of the available radium, while goethite and pyrite sorbed only 13% and 10% of total radium.

Previous work has examined radium sorption to the iron oxides, and they are typically cited as dominant sorbents in the environment. These results show some differences from previous data, particularly with the iron oxides, reporting 20-30% more sorption than found in this study (Sajih et al., 2014). Those experiments had a solid/solution ratio of 100 g/L, whereas our work had only 0.3 g/L, nearly 3 orders of magnitude in difference. This significant reduction in number of sites may explain why we observe significantly less sorption compared to others. Previous studies with sodium montmorillonite show comparable amounts of sorption for these solution conditions (Tamamura et al., 2013). Unfortunately there is little data to compare the pyrite results with, though this may be driven by the low retention of radium and difficulties of preparing experiments anaerobically, since even 1 days’ worth of air exposure can result in iron oxide coatings (MURPHY & STRONGIN, 2009).

SECTION 3.2 SURFACE COMPLEXATION MODELING

The results of the surface complexation modeling are shown in figure 2, with the fitted reaction constants in table 1.

SECTION 4: DISCUSSION/CONCLUSIONS

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