Experimental and Surface Complexation Modeling of Radium Sorption to Iron Oxies, Iron Sulfides, and Montmorillonites

Michael A. Chen\* and Benjamin D. Kocar

Dept of Civil and Environmental Engineering,

Massachusetts Institute of Technology

\*Corresponding author, [machen@mit.edu](mailto:machen@mit.edu)

**Abstract**

Radium (Ra) is a naturally occurring radioactive material that poses a significant hazard when released from the subsurface by anthropogenic activities (i.e. hydraulic fracturing), but can aslo mark groundwater movement in estuarine systems. Ra transport is known to be dominated sorption, particularly by iron and manganese (hydr)oxides, but there is limited mechanistic study of Ra binding to other common mineral surfaces. Here, we present results of batch sorption studies and surface complexation modeling of Ra to ferrihydrite, goethite, Na montmorillonite, and pyrite, in a low salinity groundwater solution at range of pH values. We find that Na montmorillonite, ferrihydrite, and goethite are major sorbents of Ra, while pyrite may play a key role during geochemical shifts common in natural aquifer systems. We also confirm currently established surface complexation models of Ra sorption to these mineral surfaces except for pyrite. Transitions in groundwater condition from anoxic to oxic will play a key role in retention or release of Ra from aquifer solids induced by redox transformations of various minerals. The results here provide useful constants and reactions that can inform modeling of Ra transport in natural aquifer systems containing many different mineral phases.

**Introduction**

Chronic ingestion and inhalation of radioactive materials, including radium (Ra) and radon (Rd), is an ongoing threat to human health worldwide.(T. Zhang et al. 2014) Of these, Ra is ubiquitous in soils, aquifers, and natural waters owing to the radioactive decay of primordial 235U, 238U, and 232Th, and often accounts for the dominant fraction of total radiation found in groundwater. All isotopes of Ra are unstable, and four (223Ra, 224Ra, 226Ra, and 228Ra) possess half-lives sufficient to persist within environmental systems and present a risk for human exposure. Moreover, 226Ra (half-life of 1600 years) is the parent radionuclide of 222Rn; chronic inhalation of 222Rn is a major driver for increased risk of lung cancer. Hence, geochemical controls on Ra mobility are directly tied to the mobility and accumulation of Rn within soil-sedimentary systems.(Jones 1999)

Several geochemical processes impart overarching controls on Ra within soils and aquifers. Alpha-recoil, the ejection of daughter radionuclides from soil and sedimentary minerals into adjacent porewater, is the primary process sourcing Ra to groundwater. Ongoing alpha recoil progressively elevates porewater Ra activities until hydrologic flushing removes the equilibrating solution, or Ra achieves secular equilibrium with its parent radionuclides. Most aquifer systems contain low but adequate (e.g. U, Th, <5 mg/kg) parent radionuclide and sufficiently favorable hydrological conditions to facilitate delivery of measurable Ra to solution.(Lu and Mason 2001) In a recent USGS study, 3% of groundwater samples (n=1270) within 7 of 15 principal US aquifers exceeded the USEPA limit for total Ra of 0.185 Bq/L.(Szabo et al. 2012) Further, high levels of Ra are often present with deeper formations, particularly shales, where low groundwater flux yield potentially hazardous activities (0.102-343 Bq/L).(Barbot et al. 2013) These naturally elevated Ra bearing formations are prevalent in some parts of the US (PA, WY, TX) and abroad (Middle East, etc.). (Lauer and Vengosh 2016; Vengosh et al. 2009) Anthropogenic activities, including uranium mining and hydraulic fracturing, can redistribute Ra and other constituents of naturally occurring radioactive materials (NORM), posing potential hazard to soils, surface waters, and aquifers.

While Ra transport is considered to be conservative in some groundwater systems, it nevertheless participates in geochemical reactions that may alter its subsurface distribution. The low solubility and rapid precipitation of Ra solids with any available sulfate generally do not constrain long term Ra behavior in most natural systems.(Fesenko et al. 2014; T. Zhang et al. 2014) Hence, Ra adsorption to solids, particularly mineral surfaces, imparts the greatest chemical control on soluble Ra transport in groundwater systems.(Gonneea et al. 2008) Under environmental conditions, Ra is not redox active, and its solution speciation is dominated by free Ra (Ra2+) across a wide range of chemical conditions (e.g. pH and salinity). Weak complexes with carbonate, sulfate, and chloride are observed, but these solution species dominate at extremely acidic or basic pH values and when ligand activities exceed environmentally relevant activities.(Grivé et al. 2015)

Although a wealth of Kd values have been tabulated for Ra adsorption to natural and synthetic solids, limited studies have used SCM to examine Ra adsorption to ferrihydrite and goethite, and only at levels far exceeding those found in most environmental systems.(Sajih et al. 2014; Sverjensky 2006) However, SCM’s have not yet been developed for Ra adsorption to reduced iron (sulfide) or clay minerals. Therefore, the objectives of this work are to 1) examine low-activity Ra adsorption to several ubiquitous minerals known or inferred to control Ra transport over a range of solution conditions found in soils and aquifers , 2) use SCM to test mechanistic descriptions of Ra adsorption to mineral surfaces, and 3) use SCM to provide quantitative comparisons of Ra adsorption to different minerals. We choose to compare sorption of Ra to ferrihydrite, goethite, sodium montmorillonite, and pyrite within a low salinity background solution and model Ra sorption behavior through SCM. These minerals are representative of widespread, dominant sorbents found in many soil-sediment systems (Na-montmorillonite) under oxic (iron oxides), and anoxic (iron sulfide—pyrite) conditions. As expected, we find that pH plays a crucial role in determining the extent of Ra sorption to most mineral surfaces; however, we also illustrate that Ra adsorption to montmorillonite is more extensive over a range of solution conditions compared to iron (hydr)oxides, which are often thought as dominant adsorbent minerals.

**Materials and Methods**

Dissolved 226Ra stock in 3% HCl was provided by the MIT Environmental, Health, and Safety office and used for all experiments. A ferrihydrite slurry and goethite powder were prepared using standard methods and added to the experiments.(Schwertmann and Cornell 2000) Powdered calcium montmorillonite STX-1b was ordered from the clay minerals society (clays.org), re-equilibrated with sodium chloride to allow for closer comparisons to previous studies of Ra sorption to Na-montmorillonites, and then cleaned of carbonates using standardized techniques.(Tamamura et al. 2013) Pyrite was ordered from Ward’s Science (www.wardsci.com), ground using mortar and pestle, passed through sieves to select for 45-250 um particles, and transferred to an anaerobic glove bag (5% H2: 95% N2: < 1 ppm O2). It was then was washed in 6 N HCl overnight to dissolve any thin iron oxide coatings, rinsed with deoxygenated DI water three times to remove residual acid, and dried anaerobically at room temperature in an open beaker. The composition of pyrite, ferrihydrite and goethite was confirmed using XRD, and surface area was measured for all minerals using BET (table S1). Further information about mineral preparation can be found in the supporting information.

Isotherms were conducted using serum vials (200 mL) filled with 100 mL of 10 mM NaCl stock solution, 30 mg of a single mineral (except for the case of pyrite, where 40 mg was used), and 5-270 Bq of 226Ra stock. Experiments using pyrite were performed in an anaerobic glove bag, and all solutions were purged with N2 prior to placement in the anaerobic chamber. The pH was titrated to 3,5,7 or 9 +/- 0.05 through use of an autotitrator, and the bottle was sealed with a thick butyl stopper. Bottles were shaken for 24 hours to allow sufficient time for sorption equilibrium.(Sajih et al. 2014) A kinetic study of Ra adsorption to montmorillonite confirmed 24 hours is sufficient to achieve equilibrium. pH was readjusted after equilibration if necessary; details on this process are in the supporting information. Acid (HCl) and base (NaOH) volume additions did not exceed 5% of the original volume. Once re-titration and re-equilibration were complete, samples were filtered using 0.22 µm PES filters, which did not sorb significant quantities of Ra. Experimental error was quantified by measuring the standard deviation of triplicates for each data point.

**Analytical Techniques.** Solutions of Ra were quantified using scintillation counting. Up to 10 mL of sample were mixed with 10 mL of Ultima Gold XR (Perkin Elmer) and sealed for 30 days to allow 226Ra to reach secular equilibrium with its daughter products. The equilibrated samples were then counted using a Beckman Coulter LS 6500 scintillation counter, and the resulting counts were compared to a calibration curve of similarly prepared 226-Ra standards to determine solution activities. This was sufficient to determine the extent of sorption and develop isotherms, with the single exception of experiments using ferrihydrite at pH 9, where gamma spectroscopy was used to quantify Ra. Details of the gamma spectroscopy and 226Ra standards are in the supporting information.

**Surface Complexation Modeling.**

Surface complexation modeling (SCM) employs mass action equations subject to thermodynamic and electrostatic constraints to describe solute-solid phase interactions. They are typically calibrated using adsorption isotherm/titration data collected across a range of geochemical conditions, sometimes with the aid of spectroscopic tools or *ab inito* modeling.(Sverjensky 2006) In general, they are used to describe and validate surface chemical reactions for specific minerals and define and estimate important surface reactions for complex mineral mixtures (e.g. soil and sediment samples). These can then facilitate predictions of solute mobility in specific aquifer systems. Challenges associated with using and comparing results of SCMs within the scientific literature include the variety of experimental systems and conditions used for model calibration, and non-standard methodology for choosing model parameters and SCM chemical reactions.(Duster 2016)

Radium binding to mineral surfaces was modeled through a double diffuse layer (DDL) surface complexation model implemented in PHREEQC.(Parkhurst and Appela 2013) Reaction formulations developed from spectroscopic measurements and used in previous studies were used to fit the experimental data.(Sajih et al. 2014; Sverjensky 2006) Details for fitting SCMs here along with some alternative modeling strategies are found in the supporting information.

**Results and Discussion**

**Sorption isotherms.** All fitted isotherms were linear within the range of activities studied, and a Kd value was calculated by fitting a line to the experimental data (Table 1). The sorption isotherm results for all minerals are plotted in figure 1. Sorption to both iron oxides show a strong dependence on pH, with ferrihydrite showing greater sorption across all pH values compared to goethite, and the extent of sorption increasing with increasing pH for both iron oxides. Differences in the surface area (table S1) explain some of the variation when comparing the extent of Ra sorption for goethite and ferrihydrite sorption, with ferrihydrite having nearly twice the surface area of goethite. Accordingly, a Ksa, defined as the Kd normalized by the mineral surface area (m2/g) (table S1), and plots of mineral sorption normalized by surface area (figure S1) are used to compare the extent of Ra adsorption between treatments (Figure S1). At circumneutral pH, goethite and ferrihydrite have relatively close Ksa values, however at more extreme values (pH = 3 and pH = 9), ferrihydrite demonstrates an appreciably larger extent of sorption compared to goethite. Further discussion of the effects of surface area and comparisons between these minerals based on surface area is in the supporting information, as Kd is a more commonly reported and used parameter. Several studies examine sorption of Ra to iron (hydr)oxides, including ferrihydrite and goethite, reporting either a K­d or data that can be used to calculate one (table S2).(Beck and Cochran 2013; Beneš et al. 1984; Gonneea et al. 2008; Sajih et al. 2014)

Two studies report isotherm data for Ra sorption to ferrihydrite, and the experimental results presented here match both reported values to within an order of magnitude of the Kd values.(Beck and Cochran 2013; Sajih et al. 2014) The K­d found in our study is the largest of the collected data sets, but was also performed at lower background ionic strength (here, 10 mM, others, 100-500 mM) and higher mineral surface area (here, 382.9 m2/g, others, ~250 m2/g) , consistent with previous results suggesting increased salinity reduces the extent of Ra sorption.(Gonneea et al. 2008) One study compared Ra sorption to hematite, ferrihydrite, goethite, and lepidocrocite, finding that ferrihydrite sorbs Ra most extensively.(Beck and Cochran 2013) This suggests the sorption isotherm results presented here represent an upper limit for Ra sorption to iron oxides in these conditions.

Reported Kd values for Ra adsorption to goethite (and experimental conditions) vary widely (table S2).(Beck and Cochran 2013; Nirdosh, Trembley, and Johnson 1990; Sajih et al. 2014) Unlike results obtained for ferrihydrite, we observe a larger extent of Ra sorption at pH 7 and 9, which may be attributed to differences in solution ionic strength and surface area of the synthesized goethite. When normalized by surface areas, KSA values are similar in some cases,(Sajih et al. 2014) but are different in others where ionic strength was much higher(Beck and Cochran 2013). Although the goethite synthesized here should more closely match those found in natural settings,(Schwertmann and Cornell 2000) other studies used different synthesis methods that often result in lower surface area. The differences between the results here and in other studies underscore the limitations of using Kd to describe and report solute-solid interactions, suggesting a more intricate understanding of surface complexes and surface sites is necessary to explain variations between similar experiments.

Sorption isotherm results for Ra onto sodium montmorillonite are plotted in figure 1, the calculated Kd and Ksa values listed in table 1. With the exception of ferrihydrite at pH 9, the total extent of sorption to montmorillonite is larger than iron (hydr)oxides over all pH values. Also, a comparatively weaker pH dependence is observed for montmorillonite sorption. This result suggests that the dominant mechanism controlling montmorillonite sorption is not complexation with pH dependent surface (edge) functional groups, but rather exchange of Ra with ions in clay interlayer. Surface area normalized isotherms also reflect these trends (figure S1). Comparison of the measured sorption Kd values here to earlier studies reveal appreciable differences, with values spanning approximately one order of magnitude (table S2). Those using a high solid-solution ratio (3000-50000 mg/L) resulted in less sorption compared to the sorption results found here, which used only a 300 mg/L solid-solution ratio.(Ames 1983; Tamamura et al. 2013) The CEC of the montmorillonites are similar to that used in this experiment (within 10 meq/100 g clay), and albeit higher here, surface areas are within a factor of three of each other, suggesting interlayer charge is also similar. These discrepancies might explain some of the differences observed,(Michael H. Bradbury and Baeyens 2005) however, a more detailed investigation into the sorption mechanisms at play in these clays, and the underlying clay features controlling will be necessary to understand the key factors controlling Ra sorption.

Pyrite showed limited sorption of Ra at low pH (3 and 5), but appreciable sorption at higher values, with little difference observed between isotherms performed at pH 7 and 9 (figure 1). The circumneutral and basic Kd values (table 1) for pyrite are comparable to that of goethite at a circumneutral pH. Unexpectedly, calculated Ksa and surface area normalized sorption values (figure S1) show that pyrite has the largest sorption of all of the minerals considered here. There is very little, if any existing data examining the sorption of Ra to any reduced iron solid. A previous study examining sorption of strontium to pyrite found no discernable sorption, contradicting the results found here.(Naveau et al. 2006) It is unclear why this should be the case, as pyrite treatment was similar in both this study and the study of strontium. The supporting information contains a more thorough examination of this discrepancy. The results here suggest that reduced iron solids will play a key role in anoxic aquifers, where dynamic variations in redox will alter the surface mineralogy of reduced iron, either releasing Ra or enhancing Ra retention. Further investigation of surface complexation reactions may elucidate the source of this discrepancy with previous work with Sr, as well as enable predictions of Ra surface complexation with pyrite under shifting environmental conditions.

**Surface Complexation Modeling**. Radium sorption to ferrihydrite and goethite was fit using two tetradentate reactions with a single site based on SCM by Sajih, 2014, which closely simulated the experimental data (figure 2). A simpler formulation that can be compared to other studies of metal sorption was also examined but had worse visual fits. The fitted reactions and constants (table 2) show that ferrihydrite and goethite have roughly similar reaction constants, but ferrihydrite has more sites, explaining why we observe more sorption extent experimentally. While the reaction constants found here are only a few log units larger than other studies’ goethite results, the constants for ferrihydrite sorption reactions here are nearly 20 log units smaller.(Sajih et al. 2014; Sverjensky 2006) Although this disparity is quite large, it is known that the structural properties (crystallinity, crystal unit size, water content) of ferrihydrite may vary substantially according to the method used for synthesis, which may account for some of the variance.(Michel et al. 2007) It is unclear though, why there would be similarities in the sorption Kd values, but such larger differences in fitted log K for surface complexation.

Concentrations of Ra used here are far below the analytical detection limits of techniques used to describe and constrain the bonding environment of Ra to solids. Hence, SCM developed here for Ra adsorption is compared with other studies that combine SCM with spectroscopic measurements, which were used to constrain surface reactions of other group 2 elements. Surface site reactions involving these other elements may have similar coordination environment and complexation reactions with solid surfaces. X-ray absorption spectroscopy was used to examine Sr binding with the surface of iron oxides, and illustrated that it forms outer sphere complexes with those surfaces.(Sahai et al. 2000) A separate study used SCM informed by x-ray spectroscopy to constrain group II element sorption to iron (hydr)oxides, suggesting Ra had tetradentate coordination with iron (hydr)oxide surface sites.(Fenter et al. 2000; Sverjensky 2006) Fitted reaction constants here, and for Sr surface complexation,(Carroll et al. 2008) however, have not closely matched the constants predicted by that study.(Sverjensky 2006) A problem that arises when comparing SCM results is that multiple reaction formulations can satisfy the constraints developed by spectroscopic observations, without a clear indication as to which is a more accurate description of sorption or if those constraints are applicable to other elements with similar chemistry.(Sverjensky 2006) These complications, and previously discussed challenges in experimental methodology for SCM, underscore the need to make measurements of Ra sorption to mineral surfaces using the same spectroscopic techniques used for other group II elements.

Surface complexation modeling of Ra adsorption to sodium montmorillonite was fit using two surface sites and an exchange reaction where Ra displaces sodium in the inner layer of the clay, following previous SCM studies of metal sorption to montmorillonites (table 2).(Michael H. Bradbury and Baeyens 2005) Fits using this model were visually good (Figure S1), and the exchange reaction explains the large Ra sorption extent over all pH values. A similar suite of reactions was used to describe Ba adsorption to Na-montmorillonite, and X-ray absorption spectroscopy confirmed the formation of both inner sphere and outer sphere complexes on the montmorillonite surface corresponding with the need for both an exchange reaction and surface site reactions in the SCM.(P. C. Zhang et al. 2001) The number of fitted sites in this model were significantly lower than reported in the literature, with literature values producing poor fits. The presence of exchange in these models account for the significant extent of sorption at acidic pHs, however, the fitted surface complexation constants in either model also suggest that Ra binds more strongly with the clay surface than either of the iron oxides.

Although SCM has not been extensively used to examine group II cation adsorption with montmorillonites, there is a broad base of literature examining the strength of exchange and surface reactions with other metals.(Mike H. Bradbury et al. 2005) Previously calculated metal exchange reactions with sodium montmorillonite cations show a range of values from 0.7 to 398, compared to 1.41 found here for Ra exchange. This suggests that Ra could easily be displaced by other metals in solution. Since a large fraction of the observed Ra adsorption is associated with the exchange reaction (i.e. sorption observed at low pH), this competition from other metals will likely play a large role in controlling Ra sorption to montmorillonites. In contrast, Ra had a relatively large surface complex reaction constants (0 and 7.5) compared to those found for other potentially hazardous metals (-20 to 2).(Michael H. Bradbury and Baeyens 2005; Gorgeon 1994) This suggests that less competition for the protonated surface sites will occur in the presence of other metals, though it is unclear how competition between the protonated surface sites and ion exchange will control the ultimate fate of Ra in high salinity environments with many competing cations. Given the large amount of sorption associated with ion exchange, we would expect Ra would be displaced by other metals in solution.

Lastly, SCM of Ra adsorption to pyrite was performed using surficial S as the adsorption site(table 2),(Murphy and Strongin 2009) based on a previous SCM that modeled Sr sorption to pyrite.(Naveau et al. 2006) Model fits of Ra sorption capture the observed data points, though not as well as for Ra adsorption to montmorillonite or iron oxides (figure S1). The fitted reaction constant is also the lowest of all of the fitted reaction constants found here by multiple log K units. This confirms the results when comparing K­d values in between minerals, yet is contradicted by the K­sa results that showed that pyrite had the largest amount of surface area normalized sorption. Increases in the total number of surface sites in the SCM were not able to fit the experimental data accurately, which suggests that this formulation of the SCM is inaccurate with respect to the actual pyrite-Ra surface complexes. Only further spectroscopic study of Ra association with a variety of pyrite surfaces can elucidate the sorption mechanisms.

**Implications for radium as a tracer in hydrogeologic settings**

The presented experimental results and SCM highlight both preferential sorption of Ra to different mineral phases present in natural aquifers, as well as the dynamic adsorption equilibria of Ra when (bio)geochemical conditions are altered, including changes in pH, salinity, and mineralogy. The presence of transformable iron minerals vs relatively stable clay minerals plays a key role in determining total Ra fluxes. The results here suggest that groundwater model predictions and estimations may improve by measuring total Ra (and in some scenarios, Ra isotopes) associated with dominant subsurface minerals, and incorporating adsorption processes into simplistic mixing models.

Radium isotopes are also used as naturally occurring environmental tracers. The mass balance of Ra isotopes in estuarine/near shore systems can provide estimates of subterranean groundwater discharge (SGD).(Moore 2003) This method matches well with other SGD measurements, but is unable to resolve groundwater behavior within estuarine systems; non-conservative mixing and retardation may occur in these zones owing to sorption or fluctuations in pH, salinity or redox state.(Burnett et al. 2002; Gonneea et al. 2008) In addition to its use as a groundwater tracer, Ra has also been identified as a marker for historic spills associated with hydrocarbon extraction.(Lauer and Vengosh 2016) This is possible if the Ra isotopic signature of produced water significantly differs from that of local groundwater, which is typical in many hydrocarbon bearing formations.(Warner et al. 2013) Successful implementation of this method requires a comprehensive accounting of Ra fate in groundwater, as transport may be significantly non-conservative due to mixing of the highly saline, often anoxic waste waters with shallow, oxic, low salinity groundwater.

Radium isotopes have played a crucial role in tracing the flux of groundwater into the ocean, and have been highlighted as a potential marker for investigating ground contamination resulting from hydraulic fracturing operations.(Lambert and Burnett 2003; Lauer and Vengosh 2016) The models used thus far are relatively simple mixing models, where transport within porous media is not considered.(Rama and Moore 1996) Study of natural Ra variations showed transport plays a critical role in controlling Ra isotope concentrations, particularly the short lived isotopes 223Ra and 224Ra, and needs more detailed models of transport to resolve these isotopes’ behavior.(Hughes, Wilson, and Moore 2015) Previous studies of Ra sorption in batch systems has provided a first basis with which to develop these models of transport, and this work contributes to these models by highlighting and comparing critical minerals that control transport, as well as providing constants and reactions to constrain Ra sorption. It also highlights areas in need of additional work to better quantify these transport processes. Investigation of Ra complexation at these surfaces, competition with redox sensitive metal ions, resolving sources of discrepancy, and further quantification of transport would be instrumental in improving Ra utility as a tracer.

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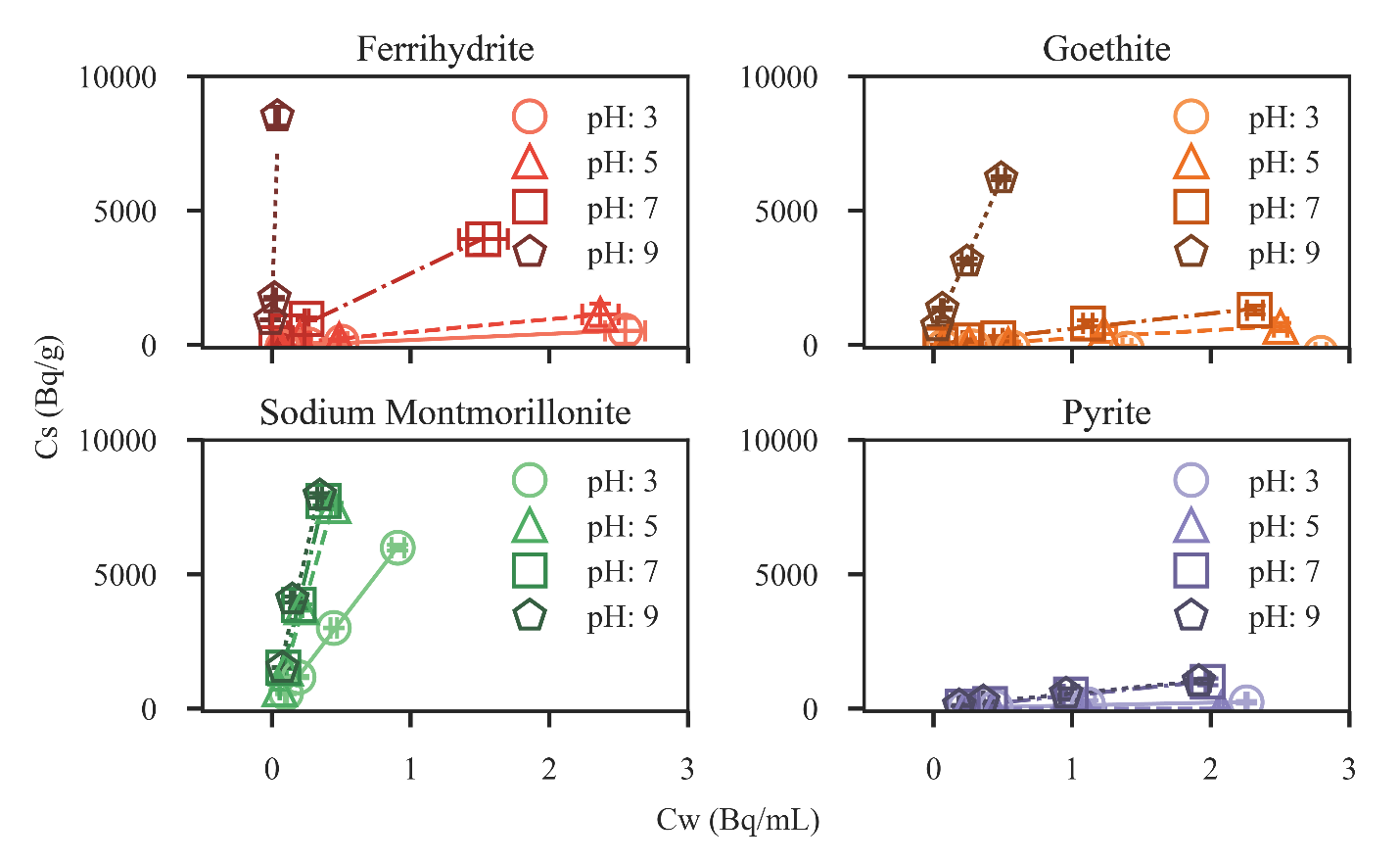
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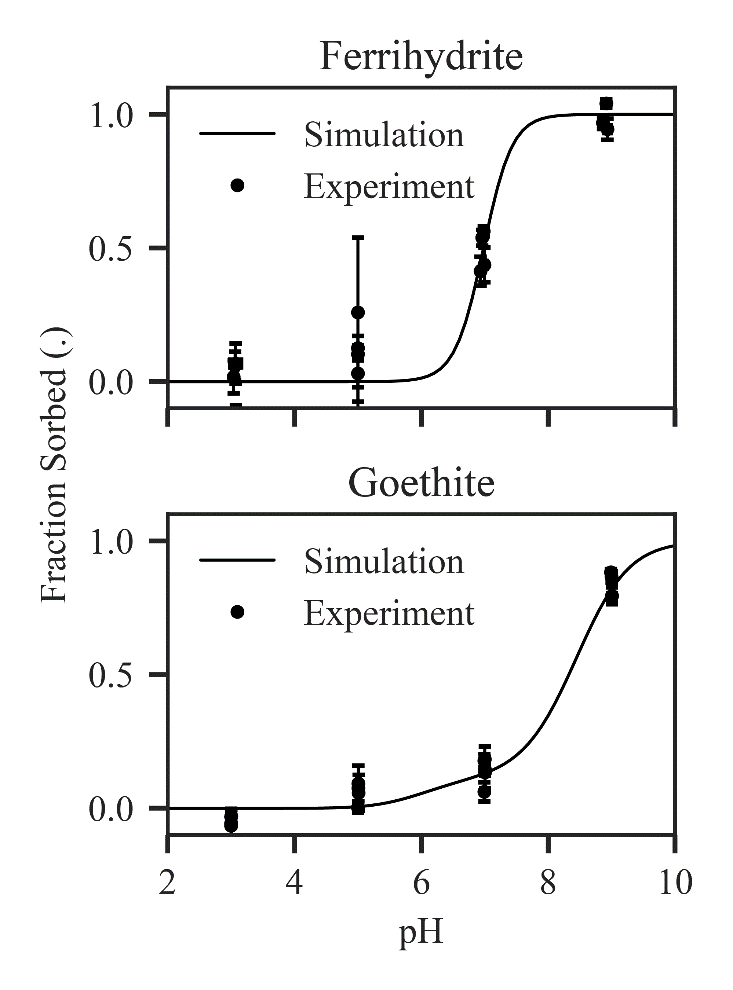
Table 1: Calculated Sorption Constants

|  |  |  |  |
| --- | --- | --- | --- |
| Mineral | pH | Kd (mL/g) | Ksa (mL/m2) |
| Ferrihydrite | 3 | 229.9 | 0.600 |
| 5 | 471.4 | 1.231 |
| 7 | 2487 | 6.495 |
| 9 | 115900 | 302.8 |
| Goethite | 3 | 0 | 0 |
| 5 | 302.7 | 2.067 |
| 7 | 573.6 | 3.917 |
| 9 | 11670 | 76.87 |
| Sodium Montmorillonite | 3 | 6740 | 134.4 |
| 5 | 17750 | 353.8 |
| 7 | 21470 | 428.1 |
| 9 | 22890 | 456.4 |
| Pyrite | 3 | 0 | 0 |
| 5 | 0 | 0 |
| 7 | 536 | 7825 |
| 9 | 520 | 7591 |

Table 2: Literature SCM reactions and fitted parameters

|  |  |  |  |
| --- | --- | --- | --- |
| Reactions | Sites (mol/g) | log K | Source |
| Ferrihydrite | 1.75E-3 |  | (Sajih et al. 2014) |
| ≡FhyOH + H+ ⇄ ≡FhyOH2+ |  | 7.92 | (Sajih et al. 2014) |
| ≡FhyOH ⇄ ≡FhyO- + H+ |  | -8.93 | (Sajih et al. 2014) |
| 4≡FhyOH + Ra2+ ⇄ ≡(FhyOH)3­FhyORa+ + H+ |  | -1.4 | Fitting |
| 4≡FhyOH + Ra2+ + 2H+⇄ ≡(FhyOH2)2­(FhyOH)2Ra4+ |  | 0 | Fitting |
| Goethite | 6.4E-5 |  | (Sverjensky 2006) |
| ≡GoeOH + H+ ⇄ ≡GoeOH2+ |  | 4.8 | (Sverjensky 2006) |
| ≡GoeOH ⇄ ≡GoeO- + H+ |  | -10.4 | (Sverjensky 2006) |
| 4≡GoeOH +Ra2+ ⇄ ≡(GoeOH)3GoeORa+ |  | -2.9 | Fitting |
| 4≡GoeOH + Ra2+ ⇄ ≡(GoeOH)4Ra2+ |  | 4.6 | Fitting |
| Sodium Montmorillonite |  |  |  |
| 2 ≡Clay-Na + Ra2+ ⇄ ≡Clay2-Ra + 2 Na+ | Exch: 8.43E-4 | 0.15 | Fitting |
| ≡ClayAOH + H+ ⇄ ClayAOH2+ | Site A: 2E-7 | 4.5 | (Mike H. Bradbury et al. 2005) |
| ≡ClayAOH ⇄ ≡ClayA- + H+ |  | -7.9 | (Mike H. Bradbury et al. 2005) |
| ≡ClayBOH + H+ ⇄ ClayBOH2+ | Site B: 6.33E-7 | 4.5 | (Mike H. Bradbury et al. 2005) |
| ≡ClayBOH ⇄ ≡ClayB- + H+ |  | -7.9 | (Mike H. Bradbury et al. 2005) |
| ≡ClayAOH + Ra2+ ⇄ ≡ClayAORa+ + H+ |  | 0 | Fitting |
| ≡ClayBOH + Ra2+ ⇄ ≡ClayBOHRa2+ |  | 7.5 | Fitting |
| Pyrite | 2.23E-5 |  | (Naveau et al. 2006) |
| ≡PyrSH ⇄ ≡PyrS- + H+ |  | 6.45 | (Naveau et al. 2006) |
| ≡PyrS- + Ra2+ ⇄ ≡PyrSRa+ |  | -10.5 | Fitting |

**Figure 1:** Isotherm results for the studied minerals. Error bars represent the standard deviation of the triplicate measurement. Fit lines’ slope matches the measured Kd values reported in Table 1.



**Figure 2:** SCM results compared against experimental results for ferrihydrite and goethite. Error bars on data points represent standard deviation of triplicate experimental measurements.