

UTILIZATION OF AMD SLUDGES FROM THE ANTHRACITE REGION OF PENNSYLVANIA FOR REMOVAL OF PHOSPHORUS FROM WASTEWATER¹

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Abstract: Excess phosphorus (P) inputs from human sewage, animal feeding operations, and nonpoint source discharges to the environment have resulted in the eutrophication of sensitive receiving bodies of water such as the Great Lakes and Chesapeake Bay. Phosphorus loads in wastewater discharged from such sources can be decreased by conventional treatment with iron and aluminum salts but these chemical reagents are expensive or impractical for many applications. Acid mine drainage (AMD) sludges are an inexpensive source of iron and aluminum hydrous oxides that could offer an attractive alternative to chemical reagent dosing for the removal of P from local wastewater. Previous investigations have focused on AMD sludges generated in the bituminous coal region of western Pennsylvania, and confirmed that some of those sludges are good sorbents for P over a wide range of operating conditions. In this study, we sampled sludges produced by AMD treatment at six different sites in the anthracite region of Pennsylvania for potential use as P sequestration sorbents. Sludge samples were dried, characterized, and then tested for P removal from water. In addition, the concentrations of acid-extractable metals and other impurities were investigated. Test results revealed that sludges from four of the sites showed good P sorption and were unlikely to add contaminants to treated water. These results indicate that AMD sludges could be beneficially used to sequester P from the environment, while at the same time decreasing the expense of sludge disposal.

Additional Key Words: acid mine drainage, residues, nutrients, water treatment.

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Introduction and Background

The release of phosphorus (P) from agricultural, industrial, and municipal sources has resulted in increased loading of P into receiving water bodies, with serious consequences for water quality and aquatic organisms (Sharpley et al., 1999; Filippelli, 2008). Therefore, sequestration of P has become of great importance, especially in sensitive watersheds, such as the Great Lakes and the Chesapeake Bay. Standard treatment technology for the removal of P from municipal wastewater involves dosing of the wastewater with coagulants such as alum [$\text{Al}_2(\text{SO}_4)_3 \cdot 18\text{H}_2\text{O}$], ferric chloride [$\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$] or lime (Metcalf and Eddy, 1991; Parsons and Smith, 2008), but the costs of these reagent chemicals often makes their use impractical or uneconomic for many wastewaters. With this in mind, various investigators have studied the use of low-cost waste products for P sequestration, such as steelmaking slag (Drizo et al., 2006), wastewater treatment residuals (Makris et al., 2004) or red muds from the processing of aluminum ores (Summers et al., 1996). Depending on local availability and transportation costs, these waste products can be economical for P sequestration. In the Appalachian region, sludges produced by the neutralization of acid mine drainage (AMD) could be especially appropriate for P sequestration. This is because AMD sludges are composed primarily of aluminum and iron hydrous oxides, the same chemical forms produced when alum or ferric chloride is added to wastewater at near neutral pH. After collection, drying and sizing, the AMD sludge residuals offer an economically attractive option for the removal of P from wastewaters (Sibrell, 2007). Several AMD sludge sources from the bituminous coal region of western Pennsylvania have previously been tested for P sequestration applications (Sibrell et al., 2009). In this study, anthracite coal mining sludges from the south central region of Pennsylvania were sampled, characterized and evaluated for their applicability in P removal technologies. Due to their close proximity to the Chesapeake Bay, these AMD sludge sources would be especially suitable for P sequestration in this sensitive watershed.

Methods and Experimental Procedures

AMD Sites and Sludge Samples

Samples of sludge produced by the neutralization of AMD were collected at six different sites in Schuylkill County, Pa., during December 2008 and May 2009 (Fig. 1). Three of the sites at the eastern edge of the county were within the Delaware River watershed, while the other

three sites were within the Susquehanna River watershed, which is the largest tributary to the Chesapeake Bay.

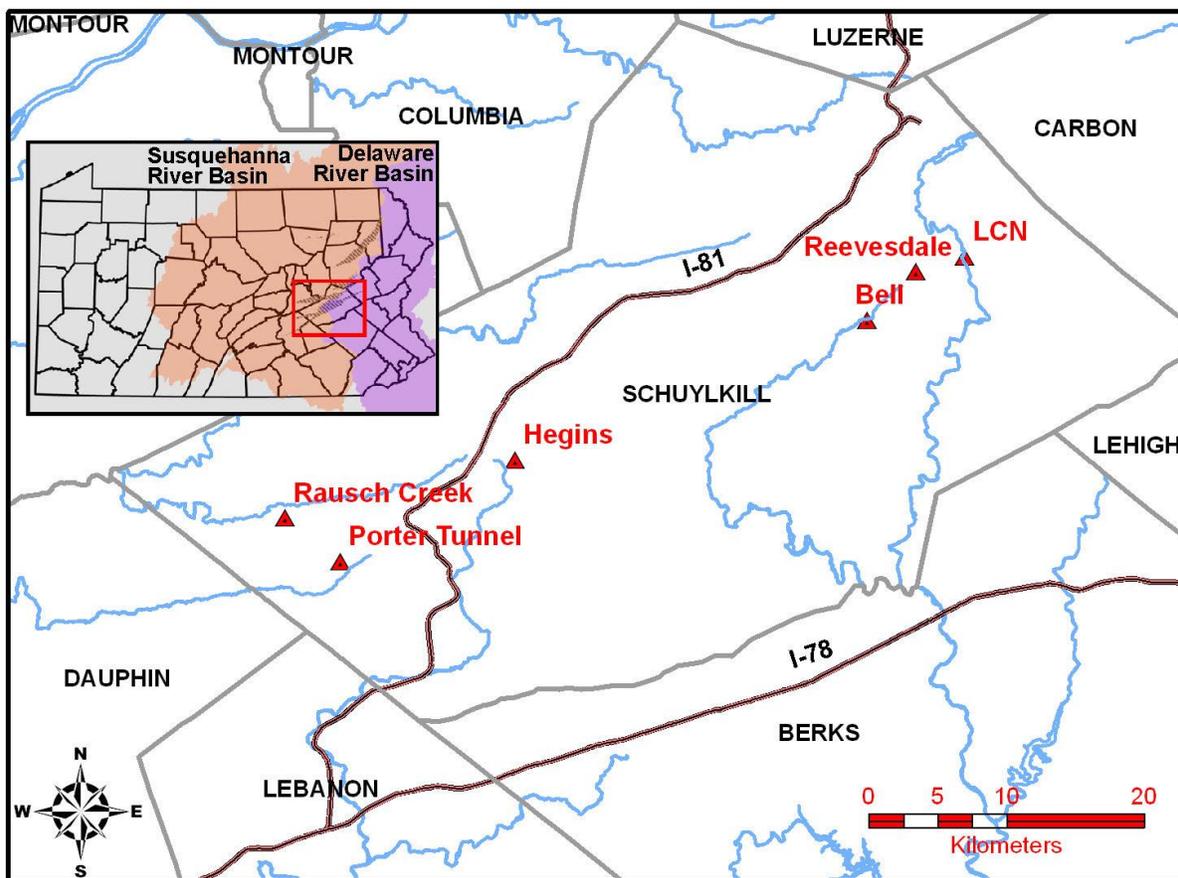


Figure 1. Map showing locations of AMD sampling points for this study.

The Rausch Creek, Porter Tunnel and LCN sites have active treatment systems, with lime neutralization, followed by settling. At the Rausch Creek plant, the sludge is further dewatered using a belt filter. In contrast, the other three sites (Hegin's, Bell and Reevesdale) are passive treatment sites, where the influent AMD is treated by flowing through limestone beds (Cravotta and Ward, 2008; Cravotta, 2008). The passive treatment sites have perforated pipe flush systems within the limestone beds for the removal of precipitated solids. The Fe- and Al-rich solids are typically flushed into a settling pond or wetland, which must be periodically dewatered and the sludge removed by mechanical means.

At the Rausch Creek and Porter Tunnel sites, samples were obtained in both December 2008 and May 2009. However, these samples are not directly comparable, because they were taken

from different locations within the plant. The 2008 Rausch Creek sample was taken from a drying bed behind the plant, while the 2009 sample was taken fresh off the belt filter at the plant. Similarly, the Porter Tunnel samples were taken from different drying beds at various degrees of moisture content. The LCN sample was obtained from a dewatering pond, where sludge from the settling pond had been pumped for further dewatering. At the passive sites, sludge samples were generally obtained by flushing the limestone bed using existing piping installed for that purpose. Sludge samples were collected by filtering the flush stream through a mesh filter bag. Different filter mesh sizes were used at some locations to explore the variability of the sludge sample collected based on the filter size opening. Additional samples were taken at the Bell site, from a wetland downstream of the limestone bed, and at Reevesdale, from washings collected by scraping ochre precipitates off limestone cobbles where the AMD influent contacted limestone at the inlet to the treatment system. Because of high water content, these latter two samples were filtered in the laboratory prior to the drying step.

Sludge samples were dried overnight at 105°C, then crushed and screened to pass through a 100-mesh screen (finer than 150 µm particle diameter). Samples were then split, and subsamples were sent to a contract laboratory for analysis. Major elements were determined by fusion with lithium metaborate, followed by inductively coupled plasma – atomic emission spectroscopy (ICP-AES). Base metals and minor elements were determined by acid digestion, followed by ICP-AES or by inductively coupled plasma – mass spectroscopy (ICP-MS), depending on element and detection limit. Carbon and sulfur contents were determined by combustion analysis.

P Sorption Capacity

Sorption capacities were measured for each of the AMD sludge sources by contacting 0.1 g of powdered solids (less than 150 µm) with deionized-water and KH_2PO_4 solutions initially containing 1, 2, 5, 10, 20 and 50 mg/L P. No buffers or ionic strength adjustments were made to the solutions, as this would not be practical for large-scale water treatment operations. The solids were weighed, added to 100 mL of solution of the desired concentration in a polyethylene bottle, then sealed and shaken at room temperature ($20\pm 2^\circ\text{C}$) on an orbital shaker for two hours at 60 rpm. Following this equilibration period, the solids were allowed to settle for approximately ten minutes, then the supernatant solution was sampled with a clean syringe, and

filtered through a 0.45- μm nylon membrane syringe filter. The filtered solution was then analyzed for P remaining in solution using a colorimetric method in accordance with Standard Method 4500-P E (APHA, 1998). The pHs of the filtered solutions were also measured for comparison with each other and with other studies.

Water Contamination Tests

The transfer of contaminants from AMD sludge media to water was measured for each of the AMD sludge samples by adding 0.1 g of powdered media (less than 150 μm) to 100 mL of deionized water in a plastic bottle, and shaking the bottle for two hours in an orbital shaker at 60 rpm. Tests were conducted at room temperature ($20\pm 2^\circ\text{C}$). After approximately 10 minutes of settling time, supernatant samples were taken from the bottles using a clean syringe, and then filtered through a 0.45- μm membrane filter. Filtered samples were placed in a plastic 50-mL centrifuge tube, acidified with two drops of trace-metal grade nitric acid, and submitted to a contract laboratory for multi-element analysis using ICP-AES. These conditions would be expected to be more severe than in a typical flow-through treatment scenario because the solids were finely divided, and the liquid solid mixture stirred thoroughly for an extended period of time. In contrast, for a typical fixed bed application, the wastewater would be passed through a static bed of relatively large particles (0.3 to 1.0 mm), with a contact time of no more than a few minutes.

Results and Discussion

Sludge Characterization

The compositions of the anthracite sludge samples are given in Tables 1a (major constituents) and 1b (minor and trace elements). Iron and Al form the majority of metal hydrous oxides in AMD sludges because they are the most common dissolved metals in coal mine drainage waters, and because both are precipitated by adjustment to neutral pH, as is typically done in treatment systems. Note that Fe and Al oxides account for more than 50% of the weight of some of the sludge samples. However, there is considerable variation, due to different influent water chemistry, treatment systems and method of collection. In contrast with Fe and Al, Mn is often underrepresented in resulting sludges because the pH at which insoluble manganese oxides form is much higher than that for Fe or Al or because, in passive treatment system, Mn is more challenging to remove than Fe or Al (Kirby et al., 1999).

Table 1a. Chemical analysis of AMD sludge samples. Major constituents by ICP-AES after lithium metaborate fusion.

Sample ID	SiO ₂ %	Al ₂ O ₃ %	Fe ₂ O ₃ %	CaO %	MgO %	MnO %	Na ₂ O %	LOI* %	Total** %	C %	S %
Rausch Creek 1	27.5	10.25	21.9	2.61	2.91	3.65	0.19	27.5	98.2	11.55	0.28
Rausch Creek 2	16.8	6.19	45.3	2.79	3.04	4.96	0.1	18.75	98.3	2.38	0.14
Porter Tunnel 1	14.0	11.4	26.6	2.9	10.7	3.38	0.1	29.1	98.5	1.26	1.11
Porter Tunnel 2	23.0	11.3	20.2	9.87	7.85	1.96	0.13	23.1	98.4	4.45	0.97
LCN	26.5	7.14	5.95	24.2	3.85	1.06	0.17	27.9	98.2	7.37	0.68
Hegins	59.2	11.4	7.46	0.18	0.84	0.99	0.29	14.0	98.3	5.39	0.21
Bell Wetland	23.3	9.84	39.3	0.21	0.19	0.05	0.19	25.6	99.9	7.93	0.76
Bell flush fine	15.8	11.5	40.2	3.07	1.24	3.56	0.15	21.8	98.2	4.18	0.73
Bell flush coarse	17.9	10.25	34.5	5.64	2.05	4.61	0.18	22.0	98.2	5.06	0.71
Reevesdale flush	7.06	23.7	36.4	0.25	0.11	0.30	0.08	29.8	98.0	2.01	1.59
Reeves. washings	12.5	15.75	20.7	16.2	2.09	0.04	0.24	29.3	98.2	5.14	0.73

* LOI indicates loss on ignition at 1000°C, and represents water, organic carbon, and other volatile substances driven off by heating.

** Numbers do not add to total shown because some minor constituents were omitted.

Table 1b. Chemical Analysis of AMD Sludge Samples. Minor Elements by ICP-AES after acid digestion.

Sample ID	As*	Cd*	Co	Cr	Cu	Hg*	Ni	Pb	Se*	Zn
	mg/kg									
Rausch Creek 1	27.6	1.7	808	40	61	0.132	1075	47	4.8	1980
Rausch Creek 2	48.1	4.7	1435	<10	119	0.039	1895	41	2.6	4390
Porter Tunnel 1	7.4	3.3	1040	10	203	0.031	1725	27	3	3450
Porter Tunnel 2	8.9	2.1	637	30	137	0.044	1035	23	2.5	2110
LCN	6.7	0.6	165	40	27	0.050	174	15	1.8	994
Hegins	10.1	0.6	567	60	101	0.069	141	94	2	224
Bell Wetland	35	<0.5	23	10	113	0.366	29	85	2.5	225
Bell flush fine	36.8	1.4	883	<10	174	0.078	494	84	2.2	1970
Bell flush coarse	34.6	1.6	1030	<10	163	0.057	654	72	2.1	2330
Reevesdale flush	58.9	0.6	66	<10	256	0.063	42	160	2.3	301
Reeves. washings	63.4	0.7	16	20	249	0.084	26	104	1.9	443
EPA Limit**	41	39	-	-	1500	17	420	300	100	2800

* Volatile elements determined by ICP-MS after digestion in aqua regia (mixture of hydrochloric and nitric acids).

**EPA maximum concentration limit for land application of biosolids.

Other common constituents of AMD sludge derive from the neutralization agent used. The LCN sample has an especially high CaO content, probably because of excess lime addition at this active treatment site. The Porter Tunnel samples show elevated CaO, as well as MgO, which suggests that the lime used at this site contains Mg. Other samples appear to have been contaminated with soils and sediments, which could have been either brought in with the influent water flow, or picked up during sampling from pond margins. The Hegins, Rausch Creek 1 and Bell Wetlands samples have high silica (SiO₂) contents probably due to inadvertent sediment contamination.

P Sorption Capacity

In order to meet the objective of effective removal of P from wastewater, the AMD sludge material should have a high affinity for P. This can be represented by preparation of a sorption isotherm for the sludge, based on the steady-state concentrations of P in the solids and solution after two hours of contact. Results for the sludges sampled in this work are given in Fig. 2, in both linear and logarithmic coordinates. The linear coordinates help illustrate the typical behavior of sorption media. At low concentrations, say below 5 mg/L in Fig. 2, the P is readily sorbed by the most of the media samples, but at elevated concentrations of P in solution, the loading of P on the solid phase begins to plateau. The plateau concentration is higher for the more effective sorption media. The pHs of the test solutions were also measured after the equilibration period, and found to range from 4.7 to 8.2. Samples containing residual CaO and MgO tended to be at the upper end of this range. Previous studies have shown that P sorption by iron oxides is relatively unaffected by pH within the range of 4 to 8 (Manning and Goldberg, 1996).

For quantification of the sorption behavior, the logarithmic form provides a better framework, as many sorption systems show linear behavior on the logarithmic plot. The Freundlich Equation is often used for sorption capacities, and assumes the following form (Weber, 1972):

$$Q_e = K_f * C_e^{1/n} \quad (1)$$

Here, Q_e is equilibrium loading of P on the solid phase, in mg/kg, and on a log-log plot is linearly related to C_e , the equilibrium concentration of P left in solution after the solid and liquid phases have reached equilibrium. On the logarithmic plot, the slope of the plot is therefore $1/n$, and the intercept is K_f . In this case, K_f can be used as a measure of the sorption capacity of the

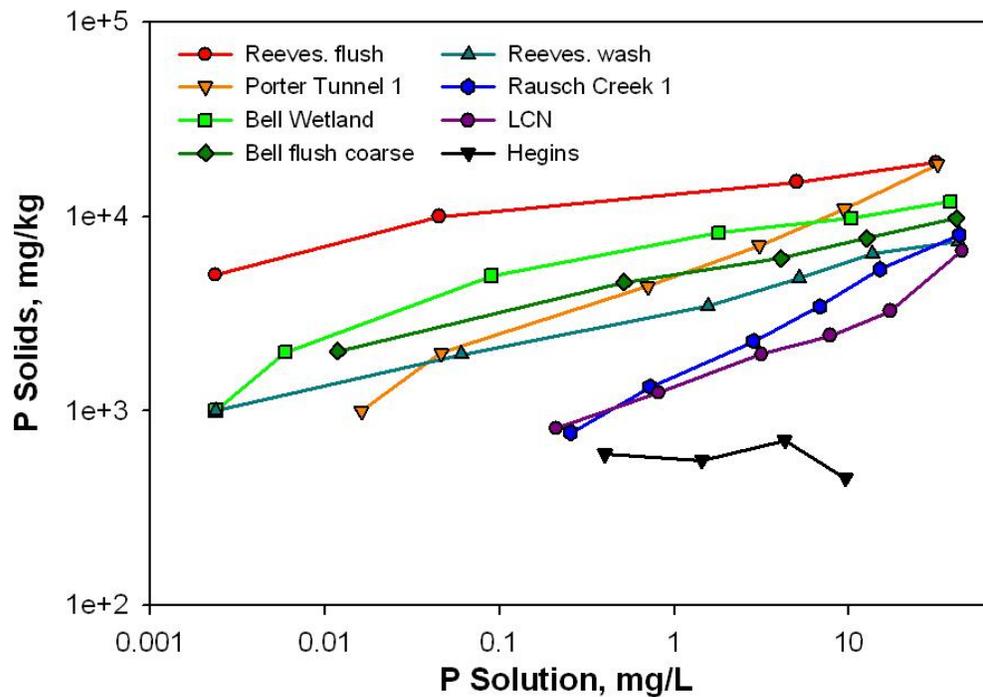
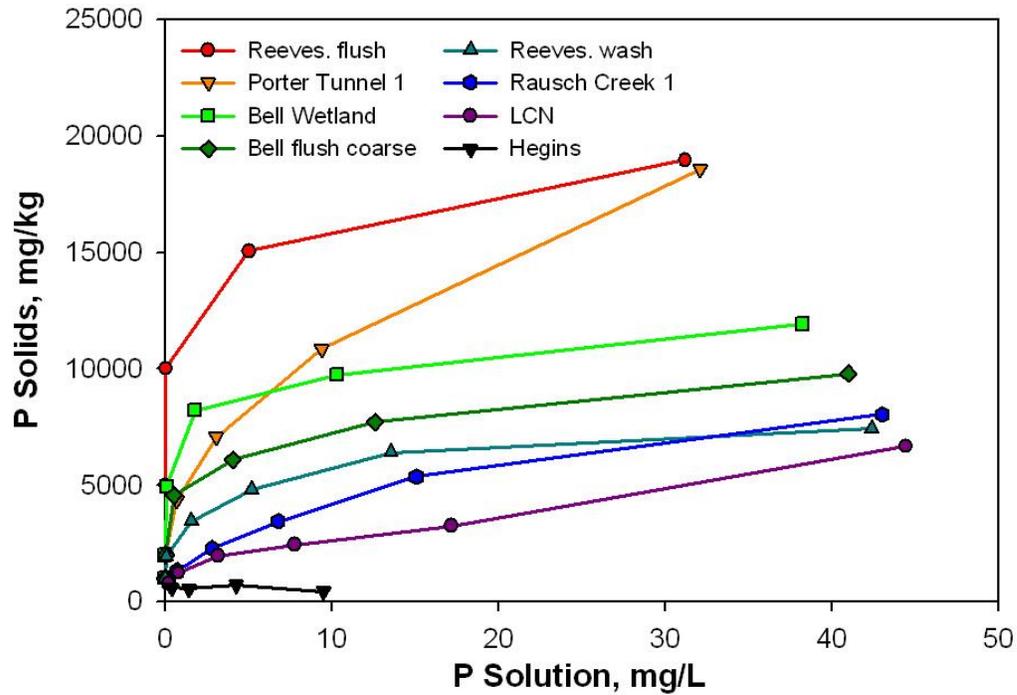


Figure 2. Concentrations of phosphorus in solids and remaining in solution for anthracite sludge samples. Tests were conducted using initial P concentrations of 1, 2, 5, 10, 20 and 50 mg/L in 100 mL solutions with 0.1 g of dried, pulverized (<150 μm) solid media shaken at 60 rpm for two hours at $20\pm 2^\circ\text{C}$. Part a: linear coordinates, Part b: log-log coordinates.

AMD sludge for P, and represents the loading of P on the solid phase when the equilibrium concentration of P in solution is 1 mg/L.

Examination of the Freundlich plots for the anthracite sludge samples reveals sorption capacities (as defined above) ranging from 600 mg/kg for the Hegins sludge to 12,500 mg/kg for the Reevesdale flush sample. For clarity, some of the sludge samples are not displayed in Figure 2, but the P sorption results for all of the samples are given in Table 2.

Table 2. Summary of P sorption data

Sludge ID	Date	Form	% Solids	K_f^*	1/n	R^2
Reevesdale flush	05.20.09	Sludge	17.2	12500	0.131	0.950
Bell Wetland	05.20.09	Slurry	22.0**	6070	0.237	0.928
Porter Tunnel 1	12.16.08	Sludge	8.7	5020	0.361	0.990
Bell flush coarse	05.20.09	Sludge	28.5	4830	0.190	0.995
Porter Tunnel 2	05.20.09	Sludge	21.5	4240	0.378	0.998
Bell flush fine	05.20.09	Sludge	26.5	3480	0.218	0.955
Reeves. washings	05.20.09	Slurry	29.9**	3450	0.207	0.995
Rausch Creek 2	05.20.09	Sludge	19.0	3090	0.414	0.998
Rausch Creek 1	12.16.08	Sludge	21.0	1470	0.457	0.998
LCN	12.16.08	Sludge	32.1	1330	0.366	0.964
Hegins	05.20.09	Sludge	32.9	600	-0.050	0.131

* K_f is the concentration of P on the solid phase (mg/kg) when the equilibrium concentration of P in solution is 1 mg/L based on the log-log plot of concentrations of P in solids and solution (Fig. 1b). For this plot, 1/n is the slope (Eqn. 1) and R^2 is the correlation coefficient.

**% Solids after preliminary filtration

The P sorption capacity for the Reevesdale flush sample is comparable to some of the AMD sludges from the bituminous area of Pennsylvania (Sibrell et al., 2009), which are being investigated for fixed bed P removal applications. The differences observed in sorption capacity between the sludges are closely related to the sludge composition, as was given in Table 1. Here, the poor sorption capacity of the Hegins sample is explained by the high silica content and the low level of Fe and Al oxides. Since the Fe and Al oxides are the active sorption materials, low levels of these oxides leads to poor sorption capacity. In contrast, the highest P sorption capacity observed was for the Reevesdale flush sample, which has low impurity content (less than 10% $\text{SiO}_2 + \text{CaO} + \text{MgO}$) and elevated Fe and Al content. P sorption capacity generally decreases as impurity content increases, and Fe and Al content decline. One caveat regarding Al content is

that some clays contain Al but would not be expected to have as high an affinity for P as the amorphous Al oxide. Clays also contain SiO₂, so a high SiO₂ content suggests that some of the Al may not be available for P sorption.

Other impurities will also cause a decrease in the P sorption capacity through dilution of the active Fe and Al oxides. For example, the LCN sample contains about 24% CaO, the highest of the tested samples. The combination of having high CaO and SiO₂ content and low Al and Fe oxide content results in a low P sorption capacity for the LCN sludge. The Porter Tunnel samples have elevated CaO and MgO contents, leading to decreased capacity relative to the Reevesdale flush sample. The Reevesdale washings sample was collected by scrubbing off the ochre that had deposited on the surface of the limestone cobbles. Here the Fe and Al oxides were diluted with CaO, probably derived from the limestone surface abraded off during the scrubbing process. The resulting sludge sample was significantly less active than the Reevesdale flush sample. The Rausch Creek 1 sample shows evidence of contamination with a high carbon content material. This may be due to coal fines carried by precipitation runoff, which often blackens the Rausch Creek sludge after rain events. Kirby et al. (1999) also observed variable and elevated C content of sludge from the Rausch Creek facility. Other samples contained lower, but still elevated levels of carbon. For those systems where limestone neutralization was used, this was probably due to inorganic carbon (calcium carbonate) contamination. For the Bell Wetland sample, the elevated carbon level was probably related to organic matter content, as some leaf and root material was noted for this sample. All of the sludge samples displayed relatively high loss on ignition (LOI) figures, which usually represent bound water, even after drying, although carbonates, sulfides and organic matter also contribute to the LOI total.

In summary, the P sorption capacity of a sludge sample is largely determined by the presence or absence of impurities that dilute the Fe and Al hydrous oxides that have a high affinity for P. These impurities can arise in many different ways, but the most common sources are un-reacted neutralization reagents and contamination of the sample with sediments.

Sludge Disposal and Water Contamination Issues

Previous studies in our laboratory (Sibrell, et al., 2009) have shown that the P can be stripped from the sludge with dilute NaOH solutions, and that the sludge can be used for repeated cycles of sorption after rinsing to remove the excess NaOH. However, it is likely that after several

cycles of sorption and regeneration, the P sorption media will be exhausted and will need to be replaced. Disposal of the spent media then becomes a possible cost consideration. Disposal options range in cost and desirability from land application to landfill disposal to hazardous waste disposal. In some cases, AMD sludge from metal mine sources has required hazardous waste disposal, due to excessive metal content and instability under landfill conditions (U.S. EPA, 2006). Sludges derived from coal mine discharges are less likely to require hazardous waste disposal, simply because of the lower concentrations of toxic metals such as Cd, Hg and Pb (Sibrell et al., 2009). Even more economically attractive than landfill disposal would be land application, as is currently practiced for some municipal biosolids wastes. Application of spent AMD ochres that have been used for P removal applications has been shown to be beneficial for plant growth (Dobbie et al., 2005), so this is an attractive disposal option once the sludge has reached the end of its useful life as a sorption media. EPA has published guidelines for impurity content in biosolids to be disposed of by land application (U.S. EPA, 1995). These guidelines are shown on the bottom row of Table 1b, and confirm that none of the anthracite sludge samples contained Cd, Hg or Pb levels above the pollution concentration limits set by the EPA for the land application of biosolid wastes. However, some of the samples from Bell, Porter, Rausch Creek, and Reevesdale treatment systems did exceed the EPA land application limits for As, Ni, and/or Zn. While these contaminant concentrations do not preclude water treatment or municipal landfill disposal, they do rule out disposal of these materials by land application. Note; however, that a 1:1:1 mixture of the Reevesdale flush, Porter Tunnel 2 and Bell Wetland sludges would meet the EPA land application pollution control limits, while still possessing an attractive P sorption capacity on the basis of average values for these samples in Tables 1b and 2. By comparison, AMD sludge samples from the bituminous region of Pennsylvania tended to contain lower levels of Ni and Zn, although As was found to exceed the EPA limit for one of the six bituminous sludges tested (Sibrell et al., 2009).

An important consideration in the use of any waste product for water treatment applications is the transfer of impurities to the water during the treatment process. Sludges deriving from the neutralization of coal mine drainage typically are better suited for water treatment applications than those from metal mines, because of the lower concentrations of toxic metals as discussed above. Each of the anthracite sludges was tested to assess the extent of contamination of water through contact with the sludge. Test conditions were more severe than would normally be

encountered through fixed bed processing, including longer contact time, more intense agitation and fine particle size of the media. Even under those conditions, the results shown in Table 3 indicate very low levels of contamination of the water from the sludge contact. Test solutions were not modified with pH buffers or stabilizers, and therefore would be expected to fall within the range of pH 4.7 to 8.2 observed in the P sorption studies. The only consistent elements present in the water samples were Ca and Mg, and these elements are typically present in tap and bottled drinking water. Also, note that the low Na and K levels, indicating the salinity of the water would not be affected by treatment with AMD-derived solids.

Conclusions

Sludge samples were collected from six different AMD treatment sites in the anthracite region of Pennsylvania for assessment of their potential for use as sorption media for sequestration of P. Sorption capacities were determined by application of the Freundlich Equation to the sorption data, and ranged from 600 mg/kg for the Hegins sample, to as high as 12,500 mg/kg for the Reevesdale flush sample. Sorption capacity was related to sludge composition, with samples containing high levels of Fe and Al hydrous oxides showing the greatest sorption capacity.

Table 3. Analysis of water samples after sludge contact.

Part a: Major sludge constituents

Sample	Al (mg/L)	Ca (mg/L)	Fe (mg/L)	K (mg/L)	Mg (mg/L)	Mn (mg/L)	Na (mg/L)
Rausch Creek 1	<1	1.9	<1	<5	1.89	0.01	<1
Rausch Creek 2	<1	1.6	<1	<5	1.88	0.01	<1
Porter Tunnel 1	1	5.1	<1	<5	4.33	<0.01	<1
Porter Tunnel 2	2	16.3	<1	<5	0.76	<0.01	<1
LCN	<1	10.1	<1	<5	0.51	<0.01	<1
Hegins	<1	<0.5	<1	<5	0.08	0.48	<1
Bell Wetland	<1	0.8	<1	<5	0.11	0.08	<1
Bell flush coarse	<1	6.4	<1	<5	0.27	<0.01	<1
Bell flush fine	<1	4.4	<1	<5	0.28	<0.01	<1
Reevesdale flush	<1	0.7	<1	<5	0.06	0.04	<1
Reeves washings	2	12.6	<1	<5	0.05	<0.01	<1

Part b: Trace metals

Sample	As (mg/L)	Cd (mg/L)	Co (mg/L)	Cu (mg/L)	Ni (mg/L)	Pb (mg/L)	Zn (mg/L)
Rausch Creek 1	<0.05	<0.001	<0.02	<0.01	<0.01	<0.05	0.02
Rausch Creek 2	<0.05	<0.001	<0.02	<0.01	0.02	<0.05	0.01
Porter Tunnel 1	<0.05	<0.001	<0.02	<0.01	<0.01	<0.05	<0.01
Porter Tunnel 2	<0.05	<0.001	<0.02	<0.01	0.01	<0.05	0.01
LCN	<0.05	<0.001	<0.02	<0.01	<0.01	<0.05	0.08
Hegins	<0.05	<0.001	<0.02	<0.01	0.03	<0.05	0.01
Bell Wetland	<0.05	<0.001	<0.02	<0.01	<0.01	<0.05	0.02
Bell flush coarse	<0.05	<0.001	<0.02	<0.01	0.02	<0.05	0.01
Bell flush fine	0.07	<0.001	<0.02	<0.01	<0.01	<0.05	<0.01
Reevesdale flush	<0.05	<0.001	<0.02	<0.01	0.01	<0.05	0.14
Reeves washings	<0.05	<0.001	<0.02	<0.01	<0.01	<0.05	<0.01

Test conditions: 0.1 g of powdered sludge added to 0.1 L deionized water, mixed for 2 h on an orbital shaker, then filtered through 0.45 µm syringe filter and acidified for analysis.

When the Fe and Al content decreased, due to contamination of the sludge with excess reagent or sediments, P sorption decreased as well. Sludge samples were also analyzed for the presence of toxic metals, as these would impact sludge disposal costs and could result in contamination of the treated water. The concentrations of Cd, Cu, Hg, Pb and Se were all below the EPA guidelines for land application of biosolids; however, some samples did exceed As, Ni and Zn limits. However, judicious blending of different sludge samples could result in a composite sample that would be suitable for land application. Results from the water contamination test showed that transfer of impurities was minimal even under the prolonged contact time and small particle sizes in contact with deionized water. These results suggest that the anthracite sludge sources should be considered an asset rather than a liability, capable of lowering costs both for AMD remediation and P sequestration, and that further development of the concept should be undertaken. Additional studies of the total contaminant concentrations, leachable concentrations, and sorption capacities of blended sludges from the sites investigated as well as other possible sources may be useful to identify optimal P sorption materials.

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