Characteristics of sludge produced from passive treatment of mine drainage

Brian A. Dempsey[1] **& Byong-Hun Jeon**[1]
[1]*The Pennsylvania State University, Department of Civil & Environmental Engineering, 212 Sackett Building, University Park, PA 16802, USA*

ABSTRACT: In the 1994 paper by Brown, Skousen & Renton it was argued that settleability and wet-packing density were the most important physical characteristics of sludge from treatment of mine drainage. These characteristics plus zeta-potential, intrinsic viscosity, specific resistance to filtration, and coefficient of compressibility were determined for several sludge samples from passive treatment sites and for several sludge samples that were prepared in the laboratory. Sludge from passive systems had high packing density, low intrinsic viscosity, low specific resistance to filtration and low coefficient of compressibility compared to sludge that was produced after addition of NaOH.

KEYWORDS: *acid mine drainage, sludge, high-density sludge, settling rate, wet packing density, zeta-potential, coagulation, viscosity, specific resistance to filtration, compressibility*

INTRODUCTION

Mine drainage often contains free proton acidity (mineral acidity) as well as high concentrations of iron and sulphate. Treatment of mine drainage can be categorized as active or passive. Conventional active treatment systems include neutralization by addition of alkaline chemicals (NaOH, Na_2CO_3, CaO and $CaCO_3$) to acidic wastewater. Passive systems may include anoxic limestone drains (ALD's) to increase alkalinity; passive abiotic ponds to transfer oxygen and to precipitate ferric oxides; anoxic biotic zones that are supplied with organic matter to remove sulphate and iron and to increase alkalinity; and aerobic wetlands or limestone filters in which the oxidation of Fe(II) and Mn(II) is catalyzed by microorganisms.

Sludge from passive abiotic ponds for treatment of mine drainage contains iron oxy-hydroxides such as goethite, lepidocrocite, and phases that convert to hematite upon heating (Kirby 1999*a*). Kirby also found gypsum, bassinite, calcite and ettringite in sludge produced during active treatment. Bigham *et al.* (1996) also reported that the dominant Fe(III)-bearing phases and the presence of additional phases depended on the pH of treatment.

Since sludge is removed annually or less frequently at passive aerobic ponds, the sludge should be as dense as possible. It is also desirable that sludge be fluid enough to be pumped and dewatered. The literature indicates that low-density sludge is formed when alkaline chemicals are added to solutions of ferric salts. Brown *et al.* (1993) added various alkaline chemicals (NaOH, Na_2CO_3, CaO, or NH_4OH) to mine drainage. All of the sludge samples had low total solids concentration (between 0.5 and 1.5%) after settling for one week. Ackman (1982) noted that sludge from active chemical treatment of mine drainage has low total solids and therefore occupies relatively large storage volumes. He stated that sludge from chemical treatment of mine drainage is difficult to dewater, and that it would be useful to design treatment that produces dense, easily handled, and readily disposable sludge. Sludge from passive systems, however, seems to be different. Hsieh (1993) showed that sludge from passive treatment of mine drainage was easily dewatered compared with ferric oxide sludge that was prepared by addition of chemical reagents.

Production of sludge from mine drainage involves both oxidation and precipitation processes. Precipitation of ferric oxy-hydroxide can occur as a new particle due to a homogeneous oxidation process (Liang *et al.* 1993). Precipitation can also occur at the interface between water and existing ferric oxy-hydroxide and this is called heterogeneous oxidation (Tamura *et al.* 1976; Sung & Morgan 1980; Sarikaya 1990; Wehrli 1990; Ames 1998). Ames (1998) concluded that most oxidation of Fe(II) in passive abiotic ponds occurred by the heterogeneous mechanism. Herman *et al.* (1984) described a high-density sludge (HDS) process that can produce 20 to 30% solids after sedimentation and 40 to 50% solids after storage. HDS processes require that precipitation occur at an interface rather than as a new particle. There are several proprietary processes for HDS. Often, lime is added to a portion of the recirculated sludge to obtain a pH of about 13. At this high pH, recirculated sludge is combined with mine drainage, coal pile runoff, or acid industrial wastes so as to adsorb Fe(II) onto the recirculated solids. Then the slurry is aerated and precipitation is at the interface rather than as a new primary particle. Dempsey (1993) and Dempsey *et al.* (1998) have reviewed and critiqued strategies that can be used to produce HDS.

It is useful to compare active treatment of mine drainage with coagulation processes in water and wastewater treatment (Dempsey 1993; Dempsey *et al.* 1998). In water and wastewater treatment, it is desirable to produce light flocculent solids with most of the surface area within a fractal aggregate. These materials have large cross-sections and are better at sweeping absorbable contaminants out of the water. These materials are intentionally formed under *active* conditions (i.e. large degree of super-saturation). However, it is difficult to consolidate and to dewater these sludges, thus treatment of residuals from water and wastewater treatment may consume half of the operating budget. Cornwell & Vandermeyden (1999) reported total solids of 1.9% for a water treatment sludge that was produced by

Geochemistry: Exploration, Environment, Analysis, Vol. **1** 2001, pp. 89–94

Table 1. *Post-ALD concentrations (mean ± standard deviation) at the HB & CK sites (from Roscoe 1999)*

Parameter (units)	HB	CK
Flow ($m^3\ d^{-1}$)	185 ± 22	618 ± 407
pH	6.18 ± 0.15	6.27 ± 0.13
Total inorganic carbon ($mg\ l^{-1}$)	74.1 ± 6.3	143.8 ± 14.4
Ferrous iron ($mg\ l^{-1}$)	245 ± 12	94 ± 11
Alkalinity ($mg\ l^{-1}$ as $CaCO_3$)	146 ± 31	309 ± 46
Net alkalinity ($mg\ l^{-1}$ as $CaCO_3$)	−320	+130
Calcium ($mg\ l^{-1}$)	211 ± 14	306 ± 29
Magnesium ($mg\ l^{-1}$)	135 ± 22	202 ± 27
Manganese ($mg\ l^{-1}$)	38	48
Sulphate ($mg\ l^{-1}$)	910 ± 36	1600
Ionic strength	0.069	0.064

addition of NaOH to ferric chloride. Elliott *et al.* (1990) reported an average of 6.6% total solids for 71 water treatment sludge samples after several years settling and consolidation in storage lagoons. Similarly, pH and dissolved oxygen are high in most active treatment systems so that the chemical driving force for precipitation of ferric oxy-hydroxides is huge, resulting in immediate formation of very small primary particles.

The passive treatment systems that were studied in this paper have anoxic limestone drains (ALD's) followed by aerobic ponds and wetlands. The focus of the paper is on solids that are produced within the aerobic ponds. The ALD's add alkalinity to acid mine drainage by dissolution of $CaCO_3(s)$. Properly designed ALD's result in net alkalinity, i.e. more alkalinity that is required to neutralize the original mineral acidity plus additional acidity that is generated by oxidation of Fe(II) and precipitation of ferric oxy-hydroxides. ALD discharge has high pH buffer capacity due to high concentrations of bicarbonate and of carbonic acid. Typically, the pH in post-ALD aerobic ponds stays between 6.3 and 6.8 due to high buffer capacity, and since decrease in pH due to oxidation/precipitation is compensated by volatilization of CO_2.

The objective of this study was to examine the characteristics of suspended and settled solids in passive aerobic ponds for treatment of ALD-treated mine drainage. Results will also apply to seeps that are net alkaline without ALD treatment. Specifically the wet-packing density, settling velocity, specific resistance to filtration, compressibility, zeta-potential and intrinsic viscosity were measured for sludge samples and solids from several field sites and after oxidation of several synthetic mine drainage solutions.

MATERIALS AND METHODS

Site details and sludge samples

Field sludge samples were obtained from two treatment sites in Western Pennsylvania, abbreviated CK (a C&K Coal site) and HB (Howe Bridge). The HB site has been studied and described by several investigators (e.g. Ames 1998; Jeon 1998; Kirby *et al.* 1999*b*; Roscoe 1999). HB and CK are passive systems, except that NaOH is added at the influent of CK Pond #1 (downstream from the channels that are emphasized in this research). CK Pond #1 is considered an active process due to the addition of NaOH and the high pH (>10 at the point of NaOH addition). All other sites were considered passive, since the pH was always <6.8.

Post-ALD chemistries of the two sites are described in Table 1. CK had net alkalinity (309 $mg\ l^{-1}$ initial alkalinity compared to 180 $mg\ l^{-1}$ that would be required to convert Fe(II) to ferric oxy-hydroxide). HB was net acid, i.e. the initial alkalinity was inadequate to neutralize all of the acidity generated by conversion of Fe(II) to ferric oxy-hydroxide.

Table 2. *Initial concentrations for the synthetic solutions (Ames 1998)*

Parameter (units)	Syn #1 (HB)	Syn #2 (SV)	Syn #3 (CK)	Syn #4 (active)
pH	6.4	6.3	6.3	8.0
Total inorganic carbon ($mg\ l^{-1}$)	58	103	145	0[1]
Ferrous iron ($mg\ l^{-1}$)	235	84	97	0
Sulphate ($mg\ l^{-1}$)	333	636	891	0
Alkalinity ($mg\ l^{-1}$ as $CaCO_3$)	139	198	281	—
Net alkalinity ($mg\ l^{-1}$ as $CaCO_3$)	−281	48	107	—

[1] Syn #4 was $FeCl_3$ neutralized by NaOH at pH 8. Some CO_2 entered during preparation.

Field samples for suspended solids and settled solids were taken sequentially from the same location. Sludge samples were also generated in the laboratory, using the chemistries that are described in Table 2 (Ames 1998). The initial chemistry ranged from net acid (Syn#1) to small net alkalinity (Syn#2) to large net alkalinity (Syn#3), which represented the post-ALD chemistries at Howe Bridge, St. Vincent's (Latrobe PA), and the C&K Coal site. Ferric oxy-hydroxide was also precipitated by addition of NaOH to a ferric chloride solution while the pH was maintained at 8. This sludge is designated Syn#4.

Analytical procedures

Sludge samples were stored and used as slurries. The pH of slurries was measured after two standard calibrations on instruments that were accurate to ± 0.02 standard units.

Viscosity was measured using a modified Ostwald viscometer (ASTM 1997). The viscometer constant was 0.3532 (dimensionless) based on water viscosity of 1.009 cp at 20°C. Vigorous shaking prior to measurement inhibited settling of the solids inside the instrument. Field samples were pre-filtered with a large mesh filter to remove large organic and inorganic detritus. Viscosity was usually measured after adjusting to the reference total solids (TS) concentration by addition of supernatant that was collected at the same site and time. In other experiments, viscosity was measured as a function of TS. Reduced viscosity was calculated using

$$[\eta] = \frac{\eta' - \eta_s}{\eta_s c} \qquad (1)$$

where, $[\eta]$ is the reduced viscosity, η_s is viscosity of the solvent (water), η' is the viscosity of the suspension, and c is concentration of solids (TS) in $g\ cm^{-3}$. Reduced viscosity has dimensions of reciprocal concentration ($cm^3\ g^{-1}$ in this study).

Settleability tests were performed using 4.4 g TS l^{-1} for the synthetic samples and 20 $g\ l^{-1}$ for the field samples. A well-mixed sample was poured into the cylinder, quiescent conditions were achieved, and sludge volume was measured v. elapsed time. Settling velocity was linear during the first eight min after quiescent conditions were obtained. Although Vesilind (1975) suggested that settling should be measured using at least 1 l cylinders, identical 100 ml cylinders were used in this study due to small sample volumes.

The wet-packing volumes were measured after 24 h of undisturbed settling. The supernatant was slowly siphoned from the top of the sludge layer. TS were measured on the remaining sludge. Wet-packing density is reported as kg l^{-1} (dry weight/wet volume).

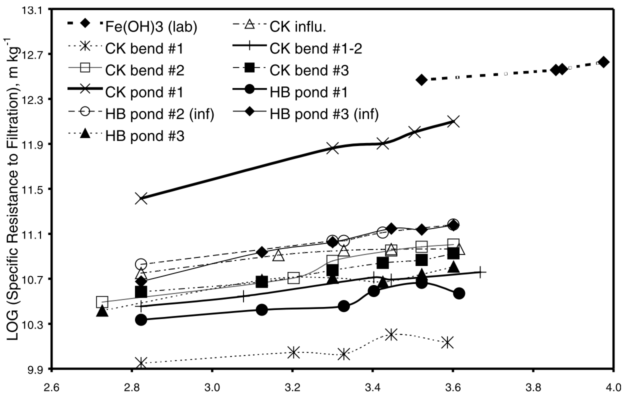

Fig. 1a. Specific resistance as a function of trans-membrane pressure for field samples.

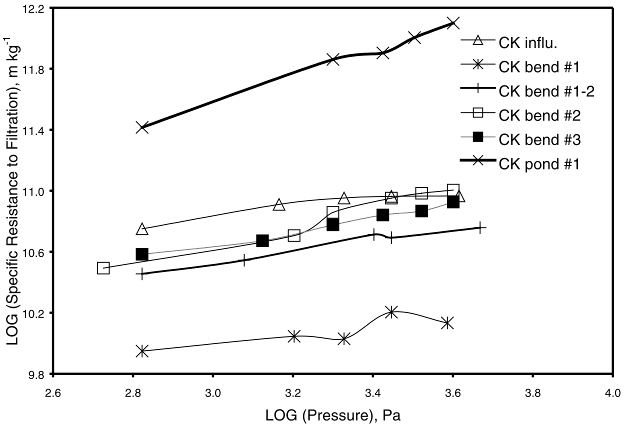

Fig. 1b. Specific resistance for CK samples.

Table 4. *Correlation coefficients (r) between variables*

Parameter	Settling rate	Wet density	Spec. resistance	S_o
Reduced viscosity	−0.900	−0.894	+0.954	+0.841
Settling rate	—	+0.891	−0.889	−0.797
Wet density		—	−0.7775	−0.838
Spec. resistance				+0.785

Zeta-potential (ZP) represents the electrical potential at the shear plane of migrating particles, usually expressed in mV. ZP is derived from electrophoretic mobility (μm s^{-1} per V cm^{-1}), which was measured using a Lazer Zee Meter™ Model 501. Samples were diluted with supernatant since the instrument is best for 10^6 to 10^9 particles cm^{-3}. The accuracy of this instrument was ±5%. The instrument was calibrated at 20°C and data were corrected for the sample temperature.

Specific resistance to filtration (SRF) was measured in a filtration cell that was equipped with a back-pressure control knob. Whatman #4 filters were used and filtering area was 0.00448 m^2. After pouring 100 ml of sludge, the unit was sealed, pressure was applied and maintained constant, and the rate of filtration was continuously measured by directing permeate into a dish on an electronic balance. Trans-membrane pressure (TMP) was 665–3990 N m^{-2} when comparisons were made among various sludge samples. The coefficient of compressibility (S_o) is the slope from plotting log (SRF) against log (TMP) (see Fig. 1).

RESULTS AND DISCUSSION

Physical characteristics of the sludge samples are shown in Table 3. The wet densities for the field samples ranged from 0.12–0.28 kg l^{-1}, except the wet density of sludge from CK Pond #1 (an active sludge) which was 0.03 kg l^{-1}. The addition of NaOH at CK Pond #1 resulted in a final pH of 7.8, although a pH of 10 was measured at the site initially. The flocculent solids from CK Pond #1 were similar in appearance to floc from potable water treatment, where it is desired to produce solids with large sludge floc volume (to sweep contaminants out of the water) and very high specific surface area (to adsorb contaminants). Sludge from CK Pond #1 was significantly different from the passive treatment sludges in most other physical parameters.

Quality of sludge from treatment of mine drainage

Specific resistance to filtration (SRF) for our passive treatment sludge samples ranged from 10^{10} to 10^{11} m kg^{-1}. Sludge from CK Pond #1 had higher SRF (to above 10^{12} m kg^{-1}) and was more compressible than sludge from passive systems. SRF of our fresh synthetic ferric hydroxides in the absence of sulphate was even higher (about 3×10^{12} m kg^{-1}).

All of the analytical parameters were related, as shown by the correlation coefficients in Table 4. Some of these relationships are illustrated in Figures 2, 3 and 4. Figure 2 shows the positive correlation between settling rate and wet density. The negative

Table 3. *Physical characteristics of the field sludge samples*

Sample	pH	Wet density (kg l^{-1})	Settling rate[1] (cm min^{-1})	Reduced viscosity (cm^3 g^{-1})	Zeta-potential[2] (mV)	SRF ($\times 10^{10}$)[3] (m kg^{-1})	S_o[4]
C&K, inlet	5.96	0.25	1.51	6.58	−11.13	5.63–10.3	0.28
Bend #1	5.86	0.28	1.93	6.64	−8.5	0.89–1.46	0.29
Bend #1-2	6.14	0.23	1.55	5.23	−10.9	2.85–5.65	0.38
Bend #2	6.43	0.14	0.85	15.71	−13.2	3.44–10.1	0.62
Bend #3	6.18	0.12	1.23	13.96	−13.3	3.83–8.43	0.44
Pond #1	7.79	0.03	0.01	43.24	−13.5	26.0–125	0.86
H.B, pond #1	6.44	0.20	1.84	7.96	−10.1	2.16–4.16	0.38
Pond #2, infl.	3.71	0.17	1.16	15.99	1.0	6.73–15.2	0.46
Pond #3, infl.	3.40	0.20	1.68	10.38	−5.5	4.73–5.0	0.65
Pond #3	2.51	0.21	1.91	11.67	1.13	3.11–6.41	0.39

[1]The settling trend of the first eight min after cessation of swirling from initial mixing. [2]Zeta-potential of solids re-suspended in supernatant. [3]Specific resistance to filtration at 665–3990 N/m^2. [4]Coefficient of compressibility.

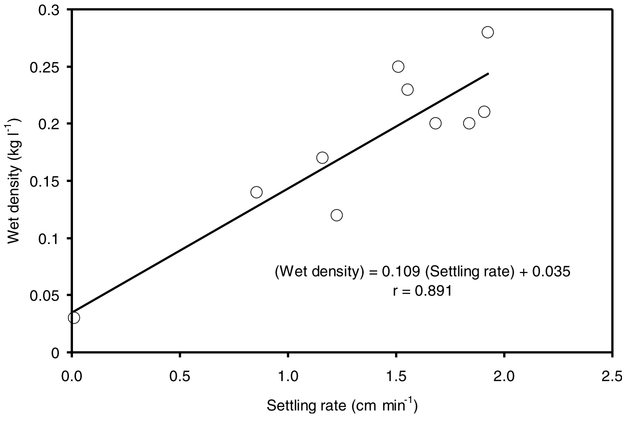

Fig. 2. Wet density v. settling rate.

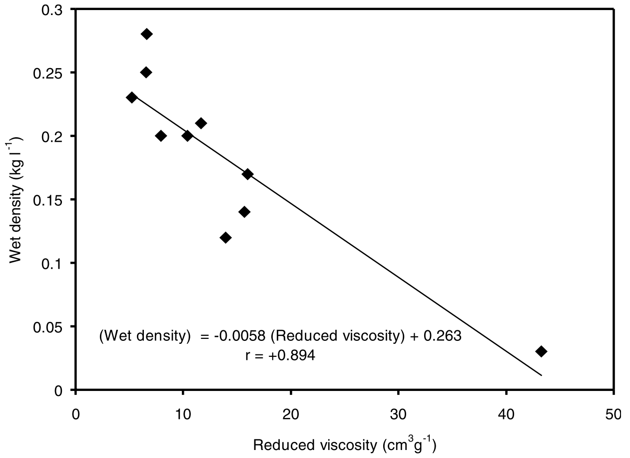

Fig. 3. Wet density v. reduced viscosity.

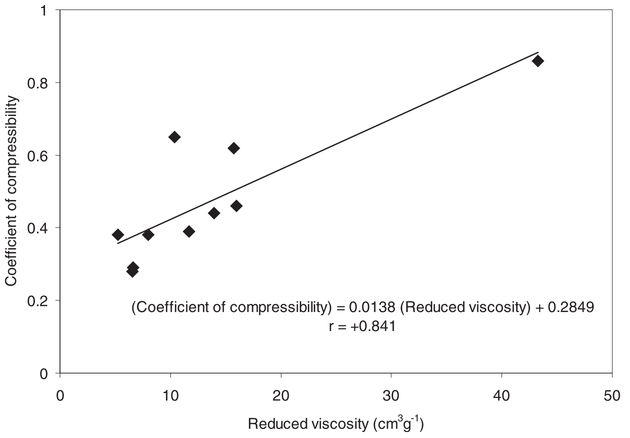

Fig. 4. Compressibility v. reduced viscosity.

Table 5. *Characteristics of the synthetic sludge samples*

Sample	Wet density (kg l^{-1})	Settling rate[1] (cm min^{-1})	Reduced viscosity (cm^3 g^{-1})	% Heterogeneous oxidation
Syn #1	0.04	0.96	31.04	60%
Syn #2	0.05	1.71	24.25	72%
Syn #3	0.07	1.89	21.98	99%
Syn #4	<0.01	ND[3]	ND[3]	0%[2]

[1]The settling trend of the first eight min after cessation of swirling from initial mixing. [2]Neutralization of $FeCl_3$ produces 100% homogenous oxidation. [3]ND=not determined. Volume of Syn #4 was insufficient for these tests.

correlation between wet density and reduced viscosity is illustrated in Figure 3. Figure 4 shows that reduced viscosity is a good indicator of compressibility.

Lower SRF is a direct indication that passive sludges are relatively easy to dewater. Low specific viscosity also indicates good dewaterability. For example, Dentel & Abu-Orf (1995) demonstrated an inverse relationship between intrinsic viscosity and dewaterability of sludge from wastewater treatment.

Novak & Calkins (1975) studied inorganic sludge samples from potable water treatment, focusing on settleability, dewaterability and ability to handle the sludge by conveyor or mechanical loader. The sludge samples came from lagoons however the conditions and time of storage were not reported. Reduced viscosities were estimated from their data as about 10 cm^3 g^{-1} for a lime sludge, 20 cm^3 g^{-1} for an iron sludge, 35 cm^3 g^{-1} for an alum-lime sludge, and about 45 cm^3 g^{-1} for an alum sludge. They observed an inverse relation between reduced viscosity and wet density, similar to our results in Table 4. Additionally, Novak & Calkins (1975) demonstrated a positive relationship between reduced viscosity and yield stress. High yield stress can be detrimental or beneficial. According to Novak & Calkins (1975), sludge can be handled by a conveyor at yield stress above 3000 N/m^2 and can be removed from a sand drying bed using a fork or a dragline at yield stress above 5000 N/m^2. Thus high yield stress is useful if one wishes to remove a low-density sludge using mechanical techniques. High yield stress would detrimental if one wished to pump sludge without dilution.

Sludge from CK Pond #1 had a high reduced viscosity and could be moved by conveyor or dragline at lower TS than the sludge samples from passive treatment (all others in Table 3). However, sludge from CK Pond #1 was difficult to dewater and had a high wet density, i.e. would have a large storage volume. Also, the higher reduced viscosity for sludge from active treatment means that the sludge cannot be pumped and piped unless TS is low. Conversely, our results showed that sludge from passive treatment of mine drainage settled well and consolidated to a high wet density. The passive sludge samples had low reduced viscosity, thus could be pumped from lagoons at high concentration.

The synthetic sludge samples (Syn#1 through Syn#4) were produced under controlled laboratory conditions, and therefore permit evaluation of the relationship between oxidation/precipitation mechanism and sludge characteristics. Syn#1, 2 and 3 were precipitated under *passive* conditions, but the different chemistries resulted in an increasing percentage of heterogeneous oxidation in the order #1<#2<#3 (Ames 1998). For Syn#4, a floc analogous to one formed in the absence of heterogeneous oxidation was expected. Some properties of Syn#1, Syn#2, Syn#3 and Syn#4 are shown in Table 5. Increasing heterogeneous reaction resulted in increased wet density (less storage volume required), decreased viscosity (easier to pump) and increased settling rates. The settling rates and reduced viscosities of Syn#1 to Syn#3 were similar to values for the consolidated passive sludge samples in Table 3, and much higher than C&K Pond #1 or Syn#4. Wet densities of the unconsolidated synthetic sludge samples were low compared to the consolidated field samples from passive systems.

Hedin (pers. comm. 1999) observed that sludge from HB ponds was more difficult to pump and dewater than sludge from other passive systems. This observation is consistent with

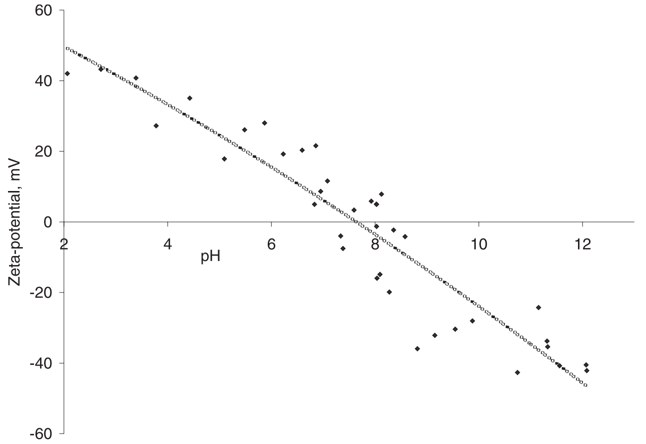

Fig. 5a. Zeta potential v. pH for synthetic samples without sulphate or Ca(II).

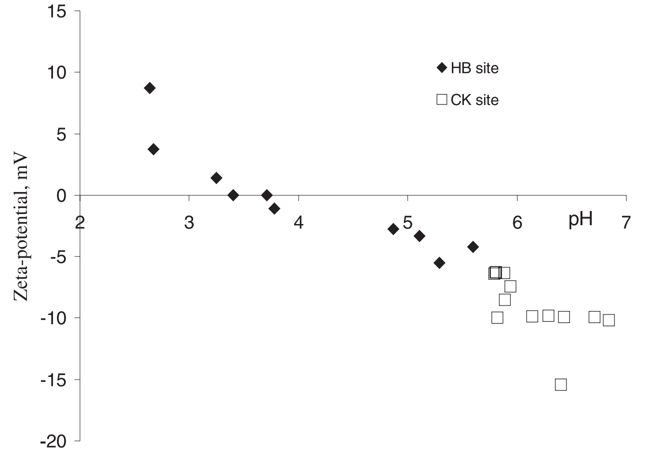

Fig. 5b. Zeta potential v. pH for HB and CK sites.

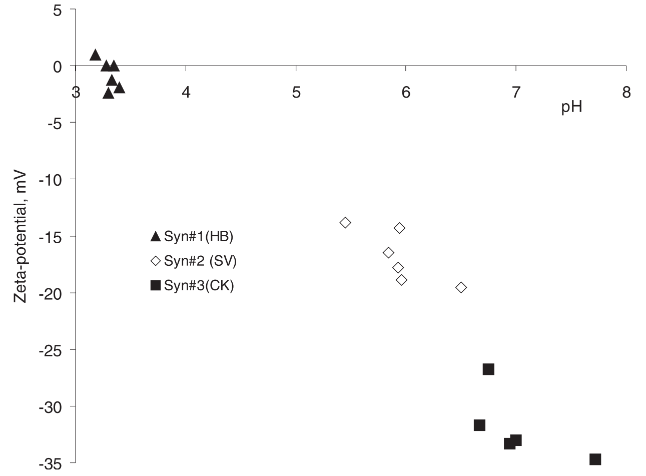

Fig. 5c. Zeta potential v. pH for synthetic samples without Ca(II).

higher reduced viscosity of passive sludge at HB compared to the CK sites, as reported in Table 3. Hedin's (pers. comm. 1999) difficulty in pumping and dewatering passive sludge from HB is consistent with Ames (1998) prediction that about 40% of ferric oxy-hydroxide at HB was produced after homogeneous oxidation of Fe(II).

Overall, the data indicates that passive treatment of mine drainage results in the formation of settleable, dense, low-viscosity and easily dewatered solids. The relationships among the various parameters indicate the quality of the sludge can be predicted and monitored, using measurements that are easy to use in the field. The data support the contention that heterogeneous oxidation results in the formation of high-quality sludge with respect to settleability, wet density, pumping and dewatering.

Coagulation and settling of particles from treatment of mine drainage

Ferric oxy-hydroxide particles that are formed during treatment of mine drainage can grow due to additional precipitation that might occur on the particle itself, or due to coagulation with other particles. Coagulation of particles can be increased by adsorption of counter-ions and by double-layer compression due to counter-ions, since these processes decrease the electrical field surrounding a charged particle, and stability of particles is an exponential function of the electrical potential (DLVO theory). Elimelech (1992) summarized the effects of counter-ions on coagulation of particles. In the absence of specific adsorption, nearly all collisions between particles will result in attachment when SO_4^{2-} approaches 500 mg l^{-1} or when Ca^{2+} approaches 200 mg l^{-1}. If specific adsorption occurs, then lower concentrations of Ca^{2+} and SO_4^{2-} will induce successful coagulation.

The pH_{iep} of laboratory-precipitated amorphous iron hydroxide was 8 in the absence of Ca^{2+} or SO_4^{2-} (see Fig. 5a) in agreement with previous research (e.g. Parks 1965; Davis 1977). The pH_{iep} of suspended solids from the field sites was 4, which indicates specific adsorption of SO_4^{2-}. Also, the range of zeta potential (ZP) values was much less for the field suspended solids (Fig. 5b) than for amorphous iron hydroxide, which is an indication of double-layer compression. Ca^{2+} was not added to the Syn#1 to Syn#3 solutions and thus the ZP values were greater for all $pH>pH_{iep}$ (compare Figs 5b and c).

The combination of high sulphate in mine drainage (which decreases pH_{iep} to about 4) and higher pH for active systems produced very negative ZP for ferric oxy-hydroxides in the absence of Ca^{2+}. This could result in poor coagulation and thus low terminal settling velocity. Permit conditions are typically written in terms of total iron. Thus a treatment process might successfully oxidize and precipitate Fe(II) but the discharge could still be in violation of water quality standards and permit conditions.

CONCLUSIONS

(1) The settled density of sludge from treatment of mine drainage (kg l^{-1} of solids) was 10 times higher for passive abiotic systems than for sludge produced by addition of NaOH.

(2) Settled sludge from passive treatment had specific resistance to filtration that was 1/10 to 1/100 the specific resistance of sludge from addition of NaOH. Also, the coefficient of compressibility for sludge from passive treatment was much lower than for sludge formed after addition of NaOH.

(3) There were good correlations between each of the following variables: intrinsic viscosity, settled density, settling velocity, specific resistance to filtration and coefficient of compressibility. This means that *in situ* measurement of viscosity or settling velocity could be used to predict subsequent behaviour of the sludge.

(4) The pH_{iep} for sludge samples from treatment of mine drainage was about 4 compared to pH_{iep} 8 for iron oxides

in the absence of SO_4^{2-}. In addition, Ca^{2+} from anoxic limestone drains kept the zeta potential (ZP) between $+10$ and -10 mV over pH of 3 to 6.5. These results indicate that particles from passive treatment systems are 'destabilized', i.e. these particles will coagulate if provided sufficient collision opportunities.

The authors thank the Ben Franklin Technology Transfer program (Commonwealth of Pennsylvania) for financial support and Hedin Environmental for technical collaboration.

REFERENCES

ACKMAN, T. E. 1982. *Sludge disposal from acid mine drainage treatment.* Bureau of Mines Report of Investigations **RI 8672**.

AMES, R. P. 1998. *Iron oxidation, gas transfer, and solids formation in passive treatment systems for mine drainage.* MSc Thesis, The Pennsylvania State University.

ASTM. 1997. *Standard test method for kinematic viscosity of transparent and opaque liquids.* D 445-96 Annual book of ASTM standards, NY.

BIGHAM, J. M., SCHWERTMANN, U. & PFAB, G. 1996. Influence of pH on mineral speciation in a bioreactor simulating acid mine drainage. *Applied Geochemistry*, **11**, 845–849.

BROWN, H., SKOUSEN, J. & RENTON, J. 1993. Floc generation by chemical neutralization of acid mine drainage. *Green Lands*, **23**, 44–51.

BROWN, H., SKOUSEN, J. & RENTON, J. 1994. Volume and composition of flocs from chemical neutralization of acid mine drainage. *Green Lands*, **24**(2), 30–35.

CORNWELL, D. A. & VANDERMEYDEN, C. 1999. Sizing residuals drying beds. *Journal of the American Water Works Association*, **91**(11), 94–105.

DAVIS, J. A. 1977. *Adsorption of trace metals and complexing ligands at the oxide/water interface.* PhD Thesis, Stanford University.

DEMPSEY, B. A. 1993. *Control of Nucleation/Crystal Growth Rates to Produce High-Density Sludges from Acid Mine Drainage and Coal Pile Runoff.* National Minelands Reclamation Center, Morgantown, WV.

DEMPSEY, B. A., BALDWIN, J., FU, F. & JEON, B. 1998. Getting rid of water: Back to the basics. *Proceedings of the 1999 WEF/AAAW Joint Residual and Biosolids Management Conference*, Charlotte NC, January 27–30, 1999.

DENTEL, S. K. & ABU-ORF, M. M. 1995. Laboratory and full-scale studies of liquid stream viscosity and streaming current for characterization & monitoring of dewaterability. *Water Resources*, **291**(12), 2663–2672.

ELIMELECH, M. 1992. Predicting collision efficiencies of colloidal particles in porous media. *Water Resources*, **261**(1), 1–8.

ELLIOTT, H. A., DEMPSEY, B. A., HAMILTON, D. W. & DEWOLFE, J. R. 1990. *Land Application of Water Treatment Sludges: Impact and Management.* AWWA Research Foundation, Denver, CO, USA.

HEDIN, R. S., NARIN, R. W. & KLEINMANN, R. L. P. 1994. *Passive treatment of coal mine drainage.* Bureau of Mines Information Circular **9389**.

HERMAN, S. T., PFEIFFER, J. B., SEWALD, R. T. & STERNER, C. J. 1984. *United States Patent, patent number 4,465,597.* Bethlehem Steel Corp, Bethlehem, PA, USA.

HSIEH, Y. 1993. Effect of ion adsorption with various hydrolysis characteristics on the dewaterability of Fe2O3 sludge. *Water Science Technology*, **28**(7), 23–30.

JEON, B-H. 1998. *The characterization of iron oxides produced from passive treatment of mine drainage.* MSc Thesis, The Pennsylvania State University.

KIRBY, C. S., DECKER, S. M. & MACANDER, N. K. 1999*a*. Comparison of color, chemical and mineralogical compositions of mine drainage sediments to pigment. *Environmental Geology*, **37**(3), 243–254.

KIRBY, C. S., THOMAS, H. M., SOUTHAM, G. & DONALD, R. 1999*b*. Relative contributions of abiotic and biological factors in Fe(II) oxidation in mine drainage. *Applied Geochemistry*, **14**(4), 511–530.

LIANG, L., McNABB, J. A., PAULK, J. M., GU, B. & McCARTHY, J. F. 1993. Kinetics of Fe(II) Oxygenation at low Partial Pressure of Oxygen in the Presence of Natural Organic Matter. *Environmental Science Technology*, **27**(9), 1864–1870.

NOVAK, J. T. & CALKINS, D. C. 1975. Sludge dewatering and its physical properties. *Journal of the American Water Works Association*, **67**, 42–45.

PARKS, G. A. 1965. The isoelectric points of solid oxides, Solid hydroxides, and aqueous hydroxo complex systems. *Chemical Review*, **65**, 177–198.

PARKS, G. A. 1967. Equilibrium concepts in natural water systems. *A symposium sponsored by the Division of Water, Air, and Waste Chemistry at the 151st Meeting of the American Chemical Society*, Pittsburgh, PA, USA.

ROSCOE, H. C. 1999. *Evaluation of passive systems for the treatment of mine drainage.* MSc Thesis, The Pennsylvania State University.

SARIKAYA, H. S. 1990. Contact aeration for iron removal: a theoretical assessment. *Water Resources*, **24**(3), 329–331.

SUNG, W. & MORGAN, J. J. 1980. Kinetics and product of ferrous iron oxygenation in aqueous systems. *Environmental Science Technology*, **14**(5), 561–568.

TAMURA, H., GOTO, K. & NAGAYAMA, M. 1976. The effect of ferric hydroxide on the oxygenation of ferrous ions in neutral solutions. *Corrosion Science*, **16**, 197–207.

VESILIND, P. A. 1975. *Treatment and disposal of wastewater sludge.* Ann Arbor Press, Ann Arbor, Michigan, USA.

WEHRLI, B. 1990. Redox Reactions of metal ions at mineral surfaces. *In:* Stumm, W. (ed.) *Aquatic Chemical Kinetics.* Wiley & Sons, NY, USA.