

# BENEFICIAL DISCHARGE OF IRON COAGULATION SLUDGE TO SEWERS<sup>a</sup>

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**ABSTRACT:** Discharge of iron coagulation sludge to sewers was demonstrated to provide multiple benefits at a wastewater treatment plant at low cost. Adding iron sludge to the collection system virtually eliminated hydrogen sulfide within anaerobic digester gas and prevented the formation of struvite throughout the solids handling facilities. Sulfides were removed via a direct reaction with the sludge, whereas struvite formation was presumably controlled by removal of soluble orthophosphate. Adding alum sludge to sanitary sewers had relatively little impact on sulfide production or struvite formation. Disadvantages of the process included elevated trace metal concentrations in the wastewater and increased solids loading.

## INTRODUCTION

Three problems at sewage treatment plants include: struvite formation in pipes, hydrogen sulfide generation in sewers and digester gas, and removal of heavy metals during sewage treatment. The removal of heavy metals from sewage is a regulatory concern, whereas struvite and sulfide formation pose operational difficulties and cause corrosion problems. This work, conducted at a 17 MGD wastewater treatment plant in Boulder, Colorado, describes an inexpensive and holistic method of addressing these issues by recycling of waste iron coagulation sludge from drinking water treatment plants.

## Struvite

The formation of struvite, an insoluble compound containing 9.9% magnesium, 7.3% ammonium, 38.8% phosphate, and 44.0% water (by weight) as  $MgNH_4PO_4 \cdot 6H_2O$ , is a major problem in many anaerobic digesters (Bogerding 1972). Although struvite precipitation leads to lowered nitrogen and phosphorus concentrations in digester effluent, the precipitate can form a scale that clogs heating and circulation lines (Pitt et al. 1992; Walker et al. 1995; Gudmundsdottir 1993). Since the total concentration of phosphorus in digesters is a function of phosphorus uptake at the treatment plant (Randall et al. 1992), problems with struvite formation might increase as phosphorus removal is improved.

Struvite formation can be controlled by acid addition, dilution, or by addition of iron salts. It has been hypothesized that ferrous iron reacts with phosphorus to form vivianite [ $Fe_3(PO_4)_2 \cdot 8H_2O$ ] solids which do not clog pipes (Pitt et al. 1992). Unfortunately, the addition of iron salts is expensive. For the Boulder facility, estimated  $FeCl_3$  costs for struvite control is \$82,000/yr based on the dose suggested by Pitt et al. 1992 (144 lbs  $FeCl_3$ /t digester volatile solids). Only \$10,000/yr is required to manually clean the struvite from the pipes.

## Hydrogen Sulfide

Methane, carbon dioxide, and hydrogen sulfide are produced during anaerobic digestion and in sanitary sewers. The methane produced at wastewater treatment plants is an important source of on-site energy; however, excessive sulfur dioxide pollutants can be produced if high concentrations of  $H_2S$  are present during combustion of digester gas. Gaseous  $H_2S$  is also very corrosive to piping, has a rotten egg odor, and causes illness or even death when present at concentrations above 50 ppm.

Gaseous hydrogen sulfides have been controlled by cleaning sewers using sulfuric acid, dosing chlorine at high concentrations, and adding iron coagulants (Walker et al. 1995; Jameel 1989; Pomeroy et al. 1946; Padival et al. 1995; Dezhm et al. 1987). It has been hypothesized (Padival et al. 1995) that soluble iron (ferrous or ferric) precipitates sulfides by formation of the following candidate solids: pyrrhotite (varies from  $FeS$  to  $Fe_4S_5$ ), ferric sulfide ( $Fe_2S_3$ ), smythite ( $Fe_3S_4$ ), and pyrite and marcasite ( $FeS_2$ ). For control of sulfides in digester gas in Boulder, assuming that a ferrous chloride dose of 29.7 lbs  $FeCl_3$ /t digester volatile solids would suffice as determined by Walker et al. (1995), chemical costs alone are \$10,500/yr.

## Heavy Metals

Although soluble heavy metals do not typically cause operational problems within domestic sewage treatment plants, they increasingly pose compliance problems in sewage effluent (Table 1). Source reduction and corrosion control are used to reduce heavy metal concentrations at treatment plants; however, these measures alone are not always adequate to meet discharge limits in the range of 2.9–20  $\mu g/L$  as is the case for copper (Moran 1994; CDPHE 1997).

Iron coagulation is a promising but expensive means of improving heavy metal removal. The coagulant directly removes particulate metals but can also remove soluble metal by coprecipitation or adsorption



where S-OH represents an adsorption site on the iron hydroxide surface. Metal hydroxide solids and recycled iron sludge can remove toxic heavy metals from wastewater solutions at  $pH > 6.5$  (Edwards 1989). In support of this general hypothesis, a recent telephone survey of 20 wastewater treatment plants suggested that overall copper removal was better at five plants dosing iron or aluminum coagulants when compared to plants without coagulation (City of Palo Alto 1994). For the Boulder facility, based on a single jar test conducted using primary influent, a  $FeCl_3$  dose of 75 mg/L is necessary to remove 59% of the copper from the influent. This dosage translates to a chemical cost of \$347,000/yr.

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**TABLE 1. Summary of Representative Metals Data at Boulder Water Treatment Plant**

Metal (1)	Average plant influent (kg/d) <sup>b</sup> (2)	Loading from Fe sludge (kg/d) <sup>c</sup> (3)	Receiving water concentration (µg/L) (4)	Plant Effluent Limit <sup>d</sup> (µg/L)		Drinking Water Limit (µg/L)	
				Concentration (5)	Composited (6)	Concentration (7)	Regulation (8)
Al	— <sup>a</sup>	8.8	440	— <sup>a</sup>	— <sup>a</sup>	50	Secondary
Cu	3.8	0.20	16	25.6	Weekly	1,300 <sup>e</sup>	Primary
Pb	0.50	0.05	4	12.2	Monthly	15 <sup>e</sup>	Primary
Mn	4.0	1.3	90	53	Monthly	50	Secondary
Zn	— <sup>a</sup>	2.5	14	218	Weekly	1,200	Secondary
Fe	120	210	870	369	Quarterly	50	Secondary

<sup>a</sup>Not available or not applicable.

<sup>b</sup>Based on 1994 and 1995 data collected after primary clarifier when ferric sludge was discharged to sewers.

<sup>c</sup>Estimates based on a single grab sample of acid digested ferric coagulation sludge.

<sup>d</sup>Calculated assuming receiving water background concentration of 0 µg/L. This limit may be lowered based on the receiving water metals concentration.

<sup>e</sup>EPA action limit.

### Study Goals

While struvite precipitation, gaseous H<sub>2</sub>S control, and heavy metal removal have typically been addressed independently, this synthesis coupled with the work of Walker et al. (1995) suggests that dosing of iron salts could simultaneously address all three problems. One main limitation is the chemical cost, which could be decreased if waste ferric coagulation sludge from drinking water treatment plants, recycled directly to sanitary sewers, could be used as the iron source. This work will: (1) demonstrate that waste iron sludge coagulation sludge can provide many of the same benefits as dosing of ferric coagulants; (2) provide mechanistic insight as to how the recycled iron sludge controls each problem; and (3) confirm bench-scale findings by thorough testing at full-scale. Economic advantages of the process were qualified and testing was conducted with alum sludge for comparison.

### MATERIALS AND ANALYTICAL METHODS

The bench-scale experimental protocol involved solution preparation, dosing of coagulant or sludge, and then measurement of sulfide, phosphorus, or metal removal. Solutions included synthetic mixtures or real waste samples collected from the 75th Street treatment plant in Boulder, Colo. The synthetic wastewater solution was designed to simulate typical bulk-ionic constituents in a digester: 5 × 10<sup>-2</sup> M NaCl, 68.4 × 10<sup>-3</sup> M NaHCO<sub>3</sub>, 20 µg Cu/L, 10 mg PO<sub>4</sub><sup>3-</sup>/L and pH 7.86 after acidification with CO<sub>2</sub> gas. Grab samples of "raw" wastewater were collected directly from the 75th Street treatment plant.

For the bench tests, a 500-mL sample of synthetic or real wastewater was added to a long-necked 500-mL Erlenmeyer flask and stirred continuously at room temperature. Synthetic solutions were deoxygenated with N<sub>2</sub> gas and a dissolved oxygen meter confirmed that oxygen was effectively excluded (<0.3 mg/L dissolved oxygen) from all solutions during the 6-hr experimental time period. For bench-scale experiments with raw wastewater, sulfides and copper were spiked into samples such that the final added concentration was 5.0 and 0.03 mg/L, respectively. Spikes were derived from appropriate stock solutions prepared from reagent grade Cu(NO<sub>3</sub>)<sub>2</sub>·2.5H<sub>2</sub>O, or NaHS·xH<sub>2</sub>O.

The prepared solutions were dosed with various concentrations of iron coagulants, freshly precipitated iron hydroxide solid, or waste ferric coagulation sludge. To represent typical coagulant dosing, iron coagulants were dosed from a 1,000-mg Fe/L stock solution. Freshly precipitated iron hydroxide solids were formed by dropwise addition of NaOH to the iron stock solution until attaining pH 9.3 and centrifugation to obtain the solids. Iron and alum sludge samples used in experiments were collected from Boulder's two drinking water treat-

ment plants, which use different coagulant types and treat different source waters. Since preliminary results indicated that 90% of the sulfide destruction during a 6-hr experiment was achieved in the first 10 min regardless of the form in which iron was added, samples were collected to assess sulfide destruction after a fixed 10-min equilibration time.

When dosing sludges, it was desirable to select doses representative of that attained through sewage discharge. An "average" dose of 0.04 mg/L Fe was calculated based on the relative flows of ferric coagulation sludge (0.08 MGD), the wastewater treatment plant flow (17 MGD), and a single grab sample of iron collected from the ferric coagulation sludge during pumping to sewers (9 mg/L). Since the sludge is discharged to the sewer on an intermittent basis (10 min every 2 hr), it was also desirable to examine higher ferric sludge levels achieved during discharge (12 times greater than the average dose). Since data on alum sludge flows are lacking, equal volumes of alum sludge were used for comparison to ferric sludge, leading to an average dose of 19.7 mg/L Al when added to wastewater. Soluble metals, operationally defined as metals passing through a 0.45 µm pore size syringe filter, are used throughout the study as an indicator of metals not likely to be removed by traditional sedimentation and filtration processes at sewage treatment plants.

Soluble cadmium, copper, manganese, and aluminum were analyzed on an inductively coupled plasma (ICP) mass spectrometer with analytical precision of better than ±0.1 µg/L for each metal (Taylor et al. 1991) and iron was analyzed on an ICP emission spectrometer with precision of ±3 µg/L according to standard method 3120 (APHA 1992). Soluble sulfides were quantified using a standard HACH colorimetric kit (methylene blue method) with a precision ±10 µg/L. Sulfides in digester gas were determined using a gas chromatograph with a PID detection with precision of ±10 mg/L. In all bench-scale experiments the removal of sulfide, phosphorus, and copper was compared to an appropriate control solution without any coagulant, iron hydroxide solid, or sludge added.

### RESULTS

Experimental results are organized in three sections. The first explores removal of sulfides by iron coagulants and recycled coagulation sludge. Thereafter, bench-scale work examined effects of sludge dosing on trace metal and phosphorus concentrations. Finally, full-scale experiences in wasting coagulation sludge to sewers are described along with implications of the work for water and wastewater utilities.

#### Role of Iron in Sulfide Removal

The efficiency of the waste iron sludge was compared to that of iron coagulants in removing a 5 mg/L sulfide spike

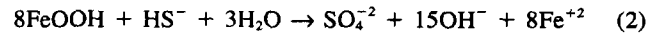
from real and synthetic wastewater. The iron coagulant was approximately twice as effective in removing sulfides as the fresh iron hydroxide (Fig. 1) in the 10-min experiment. The fact that the two sources were close in performance is surprising given that sulfide removal is thought to be via precipitation of sulfides with soluble iron species, not through reaction with an iron hydroxide solid as is present in the sludge. When the experiment was repeated using iron doses of 0.2, 0.4, and 0.8 mg/L as Fe from ferric sludge and a wastewater influent solution, sulfide removals in the range of that observed for the synthetic solutions were achieved. Although the data were not generated at comparable iron doses, sulfide destruction by iron sludge in the actual wastewater was approximately an order of magnitude more effective than in the synthetic solutions when considered on the basis of normalized iron dose (Fig. 1).

These surprising results prompted additional analysis to examine stoichiometry and to identify removal mechanisms. For sulfide removal experiments using recycled iron sludge, between 2.6 and 4.2 mg sulfide were removed/mg iron added, translating to a molar ratio of 4.5–7.2 mole sulfide/mole iron added. This is well beyond the expected stoichiometry of 1–2 mole sulfide/mole iron added if sulfide is removed by an iron sulfide precipitate.

In these experiments it was also noted that the concentration of iron passing through a 0.45- $\mu$ m pore-size filter increased significantly during the experiment. Assuming that this increase is associated with soluble ferrous iron formation and

not a small colloid, it is logical to assume that the higher soluble iron is related to sulfide removal. This hypothesis is supported by the fact that the sulfide removal during the experiment is approximately linear with soluble iron formed (Fig. 2), with about 1 mole of soluble iron formed per 43 moles of sulfide destroyed (25 mg sulfide removed/mg soluble iron formed).

The observed increase in soluble iron during sulfide removal may be the result of direct oxidation of sulfides to sulfate by iron hydroxides (Peiffer et al. 1992)



On the basis of this reaction, 8 moles soluble iron will be produced per mole sulfide removed. Of course, to the extent that the produced  $\text{Fe}^{+2}$  is subsequently consumed in a precipitation reaction with more  $\text{HS}^-$ , this experimentally determined stoichiometry would be decreased as was observed. Given the high initial levels of sulfate (132 mg/L) and this stoichiometry, no significant increase in soluble sulfate could be detected due to the preceding reaction if it occurred.

Although reported reaction rates between iron hydroxide solids and soluble sulfides can achieve the observed removals in a 10-min time period and can explain increased soluble iron during the experiment, it cannot explain the observed ratio of 4.5–7.2 mole sulfide removed/mole iron added (Peiffer et al. 1992). This prompted an investigation as to whether other constituents in the iron sludge (e.g., removed dirt and humic substances) might mediate sulfide destruction. To test this hypothesis, an alum coagulation sludge from a different treatment plant in Boulder was obtained and dosed to the influent wastewater sample at a mass concentration about 20 times higher than was previously dosed for the iron sludge (19.7 mg/L as Al). This dosage only mediated the removal of 0.8 mg/L of a 5 mg/L sulfide spike. Thus, while constituents of alum sludge are capable of removing sulfides, it also seems clear that most sulfide destruction is related to the presence of iron.

### Effect of Iron Sludge on Sorption of Soluble Phosphorus and Heavy Metals

To examine the possible role of iron sludge discharge in preventing struvite precipitation and heavy metals, jar tests were conducted to quantify removal of soluble copper, manganese, and phosphorus over the course of the previously described bench-scale experiments on sulfide removal using wastewater influent. Results are expressed based on changes in soluble metal concentrations after 30 min.

Of the 10.5 mg/L of phosphorus present in the influent, only about 0.3–0.5 mg/L was removed by dosing the iron sludge at low levels (Fig. 3) during a 30-min experiment. It is not possible to discern a significant trend given the low percentage overall removals (3–5%), but higher levels of sludge addition produced lower levels of removal from wastewater influent in this short experiment. For the alum sludge, even when dosed at 19.7 mg/L as Al, only 0.65 mg/L soluble phosphorus was removed from the wastewater influent. Thus, the alum sludge removed 0.023 mole phosphorus/mole aluminum added whereas the iron sludge removed about 0.65–4.8 mole phosphorus/mole iron added. The observed stoichiometry of phosphorus removal by the alum sludge is consistent with that expected based on sorption onto aluminum hydroxide, whereas the stoichiometry for phosphorus removal iron exceeds that based on vivianite precipitation (0.6 mole phosphorus/mole iron added).

Soluble copper, manganese, lead, and cadmium removal were also quantified after dosing of recycled ferric and alum sludges. Experiments were conducted in duplicate with less than 8% variation in results. The soluble lead and cadmium

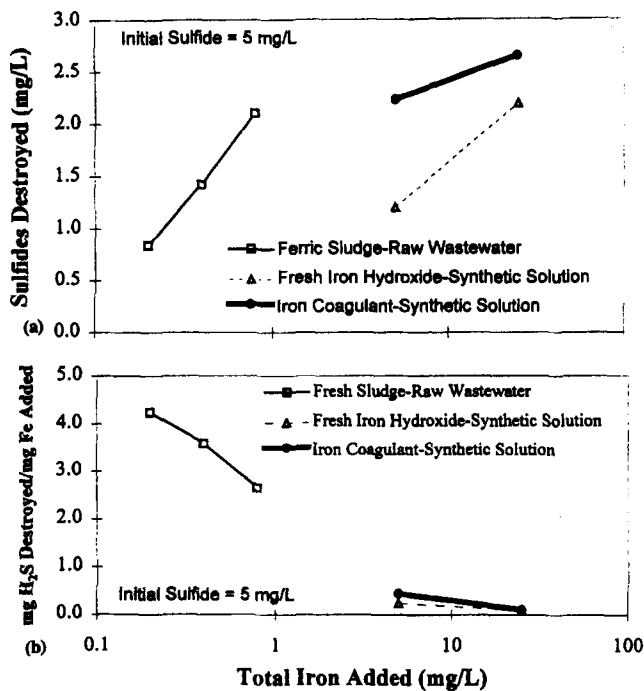


FIG. 1. Removal of Sulfides by Different Forms of Iron Added to Synthetic Solutions or Wastewater Influent

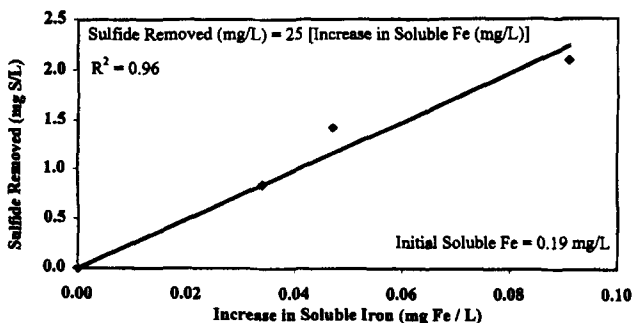
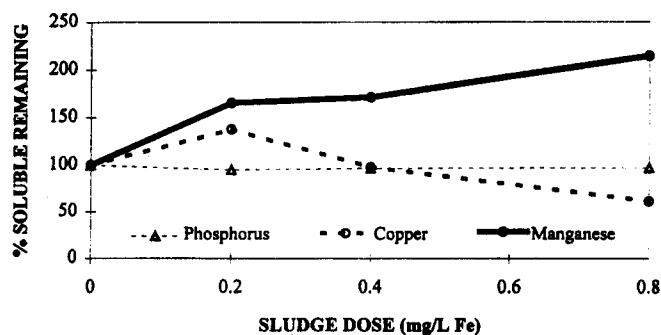


FIG. 2. Soluble Iron Formed versus Sulfide Removed



**FIG. 3. Percentage of Soluble Phosphorus, Copper, and Manganese Remaining in Wastewater Solution 30 min after Indicated Coagulant Sludge Dose—Calculated Percent Remaining =  $100\% \times (\text{Final Soluble}/\text{Initial Soluble})/\text{Initial Soluble Concentration}$ : Phosphorus = 10.3 mg/L as P; Cu = 6.3  $\mu\text{g/L}$ ; Mn = 49  $\mu\text{g/L}$**

levels were nondetectable ( $<0.1 \mu\text{g/L}$ ) in all experiments before and after sludge dosing, indicating that these metals were associated almost completely with particles. In contrast, soluble copper concentrations increased at low doses before decreasing at higher doses (Fig. 3). Soluble manganese concentrations increased at the higher doses of ferric sludge (Fig. 3). Similar trends were observed for recycled alum sludge, since high doses (39 mg/L Al) decreased soluble copper from 13 down to 3.6  $\mu\text{g/L}$  while increasing soluble manganese from 60 to 270  $\mu\text{g/L}$ .

Based on an acid digestion of each raw sludge sample, a dose of 0.4 mg/L iron sludge and 19.7 mg/L alum sludge added 25 and 176  $\mu\text{g/L}$  particulate manganese to the wastewater, respectively. Nearly all of this particulate manganese was converted to soluble manganese during the 30 min experiment described previously. In addition, digestion also indicated that the iron and alum sludges added 52.4–57.2  $\mu\text{g/L}$  total copper to each wastewater sample. The change in soluble copper concentration realized during sludge dosing depends on the relative extent of soluble copper dissolution from the sludge compared to soluble copper sorption from the wastewater. In this particular system, the net result was dependent on the sludge dose (Fig. 3).

### FULL-SCALE APPLICATION AND IMPLICATIONS FOR WATER UTILITIES

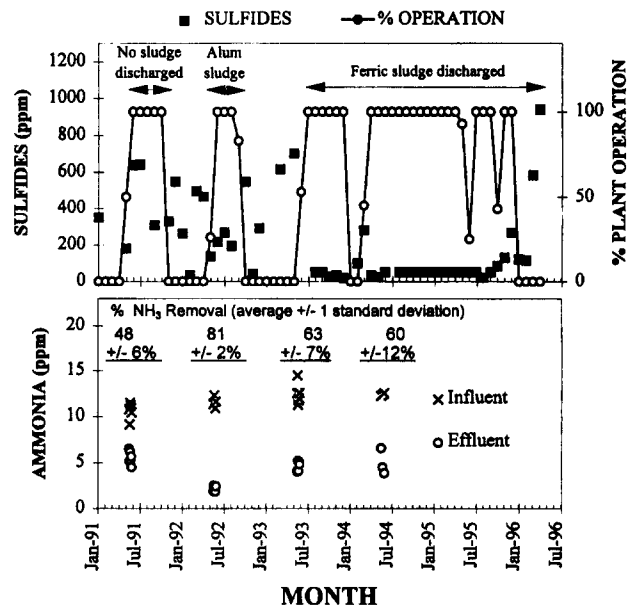
Analysis of full-scale wastewater treatment plant data confirmed the practical applicability of the previously described laboratory experiments. Boulder's drinking water is currently produced at two treatment plants, one of which operates continuously while the second operates intermittently. In the summer of 1991 the second plant was in operation without discharge of any sludge to sewers. This provided an interesting point of comparison with the summer of 1992 or 1993 when the plant was in operation while discharging only alum or only ferric sludge to the sewers, respectively (see Fig. 4; note that percent plant operation is based on the numbers of days each month the plant was in operation).

The discharged ferric sludge effectively controlled sulfides in the digester gas (Fig. 4). Sulfide levels were typically at or below detection ( $<50 \text{ ppm}$ ) after January 1993 when the treatment plant was on line. To avoid transients in data collected from digesters immediately before and after a change in sludge discharge patterns (i.e., on line versus off line), data from months in which such changes occurred were discarded leaving 41 months of data for a quantitative statistical analysis. Sulfides were present at  $401 \pm 252 \text{ ppm}$  and  $52 \pm 52 \text{ ppm}$  for months in which there was no sludge discharge and ferric sludge discharge, respectively. A high relative variation in digester gas sulfides is commonplace at this facility. While only

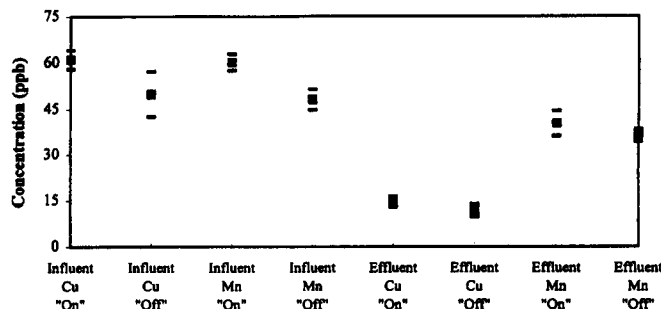
four data points were collected when the alum sludge was discharged in the summer of 1992, sulfides were  $203 \pm 54 \text{ mg/L}$  during this time. As was expected based on the bench-scale experiments, some control of sulfides might be attributed to alum sludge discharge, but not to the same extent as for the iron sludge.

Prior to discharging the iron sludge to sewers, struvite precipitation was a major problem at the Boulder facility. The struvite scale grew at a rate of between 1/4 and 1/2 in. in the centrifuge outflow pipe every 50 hr and required weekly mechanical cleaning with chisels. Moreover, struvite grew at a rate of 1/8 in. in the 600 ft of 6-in.-diameter piping associated with the centrifuge, which required contact rodding every three months. Discharging the alum sludge in 1992 did not significantly affect struvite formation.

Within a week after iron sludge discharge began in 1993, however, problems with struvite precipitation were completely eliminated. Additionally, during two time periods when iron sludge discharge was interrupted (Fig. 4), struvite precipitation problems recurred within 2–3 weeks only to completely disappear within a week after sludge discharge resumed. Rough calculations suggest that typical supplemental iron loading due to ferric discharge is about 30 kg Fe in discharge sludge/t digester dry TS. This is in the same range as the minimum 16 kg Fe/t digester dry TS (added as  $\text{FeCl}_3$ ) required to control



**FIG. 4. Effects of Drinking Water Treatment Plant Coagulation Sludge Discharge on Sulfide Levels In: (a) Digester Gas; (b) Ammonia Removal**



**FIG. 5. Mean Influent and Effluent Total Copper and Manganese for Differing Sludge Discharge Conditions—"On" Indicates Periods of Sludge Discharge; "Off" Indicates Brief Periods Without Sludge Discharge; Error Bars Indicate 90% Confidence Intervals about Mean for Two Years of Data Collected during 1994 and 1995**

**TABLE 2. Overall Chemical Costs and Expected Benefits Attributable to Various Ferric Dosing Schemes**

Operational considerations (1)	Previous: no Fe (2)	Current: Fe to sewer (3)	Digester H <sub>2</sub> S control (4)	Struvite control (5)	Heavy metal removal (6)	Sewer H <sub>2</sub> S control (7)
Estimated ferric dose	0	All ferric sludge	29.7 lbs FeCl <sub>2</sub> /t VS	100 kg FeCl <sub>3</sub> /t TS	75 mg/L FeCl <sub>3</sub>	16 mg/L Fe
Ferric dose (kg Fe/d)	0	200	51	386	1,656	1,028
Ferric feed point	None	Sewers	Digesters	Digesters	Primary clarif.	Sewers
Sulfide control in digesters reduced corrosion in co-gen facilities	No	Yes	Yes	Yes	Likely	Likely
Sulfide control in sewers	No	Some	No	No	No	Yes
Struvite control	No	Yes	Some	Yes	Likely	Likely
Heavy metal removal	No	Some possible <sup>a</sup>	No	No	Possible <sup>a</sup>	Some possible <sup>a</sup>

(a) Annual Cost Estimates for Various Options

Ferric chemical costs (\$)	0	0	\$10,500	\$82,000	\$347,000	\$210,000
Coagulation sludge disposal	\$20,000	0	\$20,000	\$20,000	\$20,000	\$20,000
Increased biosolids TS (%)	0	5	0.5 <sup>b</sup>	3.8 <sup>b</sup>	16 <sup>b</sup>	10 <sup>b</sup>
Extra biosolids handling						
Centrifugation	0	\$10,000	\$1,000	\$7,500	\$32,000	\$20,000
Land application	0	\$12,000	\$1,200	\$9,000	\$39,000	\$24,000
Co-gen H <sub>2</sub> S Impacts	\$26,000	0	0	0	0	0
Manual struvite control	\$10,000	0	\$10,000	0	0	0
Reduced concrete corrosion of sewers	0	Site dependent	0	0	0	Site dependent
<b>Total annual costs</b>	<b>\$56,000</b>	<b>\$22,000<sup>c</sup></b>	<b>\$42,500</b>	<b>\$118,500</b>	<b>\$435,000</b>	<b>\$274,000<sup>c</sup></b>

Note: Boulder cost basis—total wastewater and sewage flow = 17 MGD; digester VS = 17,300 lbs/d; digester VS = 0.77 digester TS; FeCl<sub>3</sub> = \$0.090/lb; FeCl<sub>2</sub> = \$0.110/lb; Fe(OH)<sub>3</sub> as coagulation sludge is 40% of TS in coagulation sludge.

<sup>a</sup>Possible based on literature review and if cleaner source of coagulant chemical can be located.

<sup>b</sup>Assumes that 40% of coagulation sludge TS is Fe(OH)<sub>3</sub>, assumption that coagulant addition does not significantly increase plant TS removal, and observation that wasting 200 kg Fe/d of ferric coagulation sludge increased biosolids loading 5%.

<sup>c</sup>These total benefits estimates are conservative due to possible reductions in concrete corrosion.

struvite formation at the San Francisco Water Pollution Control Plant (Pitt et al. 1992).

When the second water treatment plant was in operation, slightly higher concentrations of Cu and Mn were observed in the wastewater treatment plant influent (Fig. 5). The source of this problem was later determined to be the ferric coagulant itself, which contained 4 mg/L of copper and 46 mg/L Mn. Under typical plant operating conditions, these contaminants account for approximately 70% of the wastewater's influent manganese and 7% of the copper. Regulated levels of copper and manganese in sewage effluent are fairly stringent; in fact, for copper the regulated level is 50 times lower for the sewage effluent than for drinking water (Table 1). This leads to the ironic finding that, due to use of a relatively contaminated drinking water coagulation chemical, wastewater treatment plant compliance can be adversely affected. Future work at the Boulder facility will attempt to identify cleaner sources for the ferric coagulant chemicals—it is hoped that switching to such chemicals will actually lead to reductions in effluent metals as discussed in the literature review. Concerns over contaminated coagulants may have widespread implications for effluent metals compliance at other wastewater treatment plants, since Crozes et al. (1996) recently found 10–160 mg/L of soluble manganese in six ferric coagulants tested.

Finally, operating data from May 15–31 in 1991–1994 were compared to assess the effects of no sludge, alum sludge, and ferric sludge discharge on BOD removal and nitrification at the plant. No noteworthy trends in BOD removal were observed. The overall removal of ammonia improved from 48 ± 6% to 81 ± 2% comparing years without sludge discharge and alum sludge discharge, respectively (Fig. 4). When ferric sludge was discharged in 1993 and 1994 overall ammonia removal averaged between 60–63%. The possible enhancement of nitrification by discharging coagulant sludge should be examined in future research; however, at this time it seems appropriate to conclude that the sludge discharge did not adversely effect nitrification.

An overall cost-benefit analysis is illustrative (Table 2).

When compared to costs of traditional coagulation sludge disposal, recycling the iron resource by discharge to sewers costs the city \$2,000/year in extra biosolids handling (column 2 versus column 3 in Table 2). However, sludge recycle also saves \$10,000/year in labor to mitigate struvite precipitation and \$26,000/year for reduced maintenance of co-gen facilities. Thus, without considering the additional revenue from 2,000 extra hours of electricity production annually due to reduced downtime, the net benefit of recycling the waste ferric coagulation sludge to sewers is \$34,000/year at this 17 MGD facility.

The process is also very attractive when compared to traditional control of digester H<sub>2</sub>S or struvite precipitation using fresh ferric coagulant. Estimated savings of using recycled ferric sludge are between \$20,500–\$96,500/year without considering capital costs for coagulant storage and dosing facilities (Table 2). The possible control of sulfides in sewers may also be important, since concrete corrosion is a multimillion dollar problem at many utilities (Padival et al. 1995). If the recycled sludge prevents sulfide production in sewers as it did in the anaerobic digesters, reduced losses due to concrete corrosion are anticipated and potential savings in ferric chemical costs would be high (column 3 versus column 7 in Table 2). This possible advantage of ferric sludge recycle deserves additional future study.

Finally, in response to forthcoming enhanced coagulation regulations in the United States, many drinking water utilities will soon be forced to reconsider the basic selection of alum versus ferric as coagulant chemical to improve removal of organic matter from water supplies. Since ferric coagulants are already known to have some advantages over alum in removing organic matter (Edwards 1997), the possibility of using the waste ferric sludge as a resource for the wastewater treatment plant should be considered directly in that evaluation. At present only about 14% of all surface water treatment plants use ferric salts as coagulants (AWWA 1992), but this percentage may increase significantly given the proposed TOC regulation.

## CONCLUSIONS

Discharge of iron coagulation sludge to sewers provides cost effective control of hydrogen sulfide and struvite in digesters. An alum coagulation sludge did not provide these advantages to the same extent as the ferric sludge.

The iron sludge appears to remove sulfide by a combination of direct oxidation (to sulfate) and precipitation, as opposed to the previous assumption of precipitation alone. Iron solids in coagulation sludge seem to be as effective as new iron coagulants in sulfide removal and struvite control. Therefore, discharge of iron coagulation sludge to sewers is a convenient and inexpensive means of recycling an iron resource.

Contaminant metals present in ferric coagulant chemicals can significantly increase influent copper and manganese loading to wastewater treatment plants. This possible problem must be considered whenever use of ferric coagulants or waste ferric sludge is considered.

The process does not appear to adversely affect BOD removal or nitrification.

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