

**PROPERTIES OF WASTE RESULTING FROM  
ARSENIC REMOVAL PROCESSES IN DRINKING WATER TREATMENT**

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**ABSTRACT**

C.H. Itle

The arsenic maximum contaminant level (MCL) for drinking water is likely to be lowered sometime in 2001 or 2002. If the MCL is lowered, it is speculated that there will be stricter limits imposed on the disposal and handling of arsenic-containing residuals. The purpose of this study is to determine the properties of drinking water residuals, including the chemical characteristics, the amount of arsenic that leaches in common residual handling and dewatering processes, and the hazardous potential of the residuals.

Residual samples were collected from seven utilities with high arsenic concentrations in their raw water. Included in the study were four plants that coagulate with ferric chloride, two with aluminum hydroxide, and one softening plant. The residuals from each facility were acid digested and chemically characterized using ICP-AES to determine the total arsenic, aluminum and iron present. TCLP and California WET were performed to verify if the residuals were hazardous.

Simulated lagoons were set up and monitored over a six-month period to determine the amount of arsenic that was leached to the liquid portion over time. Toxicity testing was performed on the residuals at 20% and 100% after two months and six months of storage. Bench-scale sand drying beds were used to dewater residuals, and the leachates were analyzed to determine if arsenic leached from residuals.

The residuals were found to contain high levels of arsenic during the chemical characterization. However, all of the facilities passed the current TCLP limit of 5 mg/L. For all residuals, the California WET values were much higher than the TCLP values. In the lagoon study, redox potentials decreased and the arsenic and iron concentrations in the liquid portions increased. In the sand drying beds, very little arsenic leached; arsenic concentrations in the leachate were less than 10 ppb. Lagoon storage may not be a safe alternative for residuals containing arsenic. However, sand drying does not appear to present any threats.

There were differences between the toxicity tests performed at 100% solids and 20% solids in the lagoon study. Some of the values increased and others decreased. Additionally, toxicity testing conducted after lagoon aging differed from earlier toxicity testing, due to the changing soluble arsenic. These discrepancies suggest that the test results can be affected by the percent solids and age of the residuals, and specific instructions should be given for consistent residuals testing.

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## LITERATURE REVIEW

### *Introduction*

For centuries, large doses of arsenic have been known to be a human poison. In recent times, low doses of arsenic have been causing the most concern. The United States Environmental Protection Agency (EPA) declared arsenic a Class A Carcinogen. It has been linked to skin, liver, lung, kidney, and bladder cancer. The exposure to arsenic in drinking water may be as dangerous as the exposure to tobacco smoke and radon in homes (Smith *et al*, 1992). The Maximum Contaminant Level (MCL) for arsenic in drinking water is currently set at 50 µg/L. At this level, the risk of cancer over a lifetime consumption of 1-liter per day is 13 in 1000 (Smith *et al*, 1992).

Arsenic is a naturally occurring element in the environment. It is the fifty-second most common element in the earth's crust (Peters, 1996). It can be disturbed during mining practices and released into the environment; it can also be added to the environment through mining processes or the use of pesticides and herbicides. Arsenic is ubiquitous in our surroundings; it seems that some level of arsenic in drinking water is inevitable. The question remains as to what level is acceptable.

The 50 mg/L MCL has been under debate for several decades. The first studies linking low levels of arsenic in drinking water to cancer were published in 1967 and 1968 in Taiwan (Pontius, 1994). Beginning in the early 1980's, the EPA had begun reviewing all of the available arsenic risk assessment data to justify changing the arsenic limit. The 1996 Safe Drinking Water Act amendments called for a new arsenic rule by January 2001. Because of insufficient epidemiological data and the cost benefit analysis of imposing such regulations, the deadline for implementing a lower arsenic limit has been extended several times.

Over the past several years, much work has been done to quantify the arsenic problems in drinking water, identify treatment options, and determine costs of a lower arsenic MCL. Several review documents summarize and discuss these studies and can be consulted for further information. Included are several documents published by the EPA, such as *Arsenic in Drinking Water Rule Economic Analysis*, *Regulations on the Disposal*

*of Arsenic Residuals from Drinking Water Treatment Plants, and Technologies and Costs for Removal of Arsenic from Drinking Water* (EPA 2000a, EPA 2000b, and EPA 2000c).

### ***Disposal of Residuals Containing Arsenic***

As drinking water treatment plants remove arsenic from their water, they produce arsenic containing residuals. There is speculation that a lower arsenic MCL will impose stricter limits on the handling procedures and disposal of arsenic containing residuals. Drinking water treatment plant residuals may contain concentrated amounts of arsenic. Regulations that are more stringent may make it very difficult and expensive to handle and dispose of the waste the treatment plants generate. Very few studies have been published that characterize the amount of arsenic present in drinking water treatment residuals or the toxicity of the residuals. The characterization and toxicity testing of the residuals are important, because disposal options are ruled by these two properties.

There are several disposal alternatives available for arsenic containing residuals from drinking water treatment facilities. The options include discharge to sewers, land application, and landfilling. Because arsenic concentrations in the waste may be high, discharge to sewers is not usually acceptable for arsenic containing wastes, so this discussion will concentrate on land application and landfilling practices.

Land application is limited by the amount of arsenic present in the residuals, as regulated under 40CFR Part 503 (Edwards, 2000). Residuals containing less than 41 mg/kg arsenic can be applied to land with no limitations, while residuals containing 75 mg/kg arsenic or less can be applied in limited quantities. Land disposal is the placement of residuals into specified sludge landfills called monofills. The limit for land disposal is 73 mg/kg arsenic. The amounts of arsenic that are permissible in residuals for land disposal practices are summarized below:

Land Disposal Arsenic Limit	73 mg/kg
Land Application Arsenic Ceiling Limits	75 mg/kg
Land Application Clean Sludge Arsenic Limit	41 mg/kg

**Table 1-1.** Arsenic limits for land disposal of arsenic containing wastes, from Edwards, 2000.

Another disposal option for residuals is placement in municipal solid waste landfills. Only non-hazardous materials can be disposed of in municipal solid waste landfills. The hazardous characteristic of most concern for arsenic containing wastes is the toxicity of the waste. Presently, this is nationally characterized by the Toxicity Characteristic Leachate Procedure (TCLP), which was implemented in 1986 under RCRA. The TCLP test combines waste with an acetic acid solution and mixes it for 18 hours. The resulting TCLP extract is filtered, acid digested, and analyzed. The purpose of the TCLP is to simulate the worst-case scenario of a landfill. The test results should determine if the waste could safely be landfilled without fear of contaminating the soil or groundwater. If the TCLP extract has lower than 5 mg/L arsenic, the waste is considered non-hazardous. The TCLP value of 5 mg/L is derived from the 50 µg/L MCL; for drinking water standards, the TCLP values are 100 times the MCL. If the drinking water MCL is changed, the acceptable TCLP limit may also be changed.

Critics of the TCLP test contend that it underestimates leachate from some high alkaline wastes (such as softening residuals) because of the buffering capacity of the wastes (EPA, 1996). The TCLP test may also not mimic conditions in non-hazardous industrial landfills. The chelation-facilitated mobility of some wastes is underestimated by the TCLP because of the low chelation activity of the TCLP acetate buffer. Additionally, the TCLP does not account for oxidation/reduction reactions that occur in landfills. This is probably its most serious failure since arsenic is more soluble under reduced conditions.

One study declared that the TCLP was not a close simulation of landfill conditions because it failed to extract oxoanion- forming elements such as arsenic (Hooper *et al*, 1998). Oxyanion forming elements combine with oxygen to form negatively charged solutes that will not complex with the negatively charged acetate solution used in TCLP extracts. In Hooper's 1998 study, TCLP extracts and actual municipal solid waste leachate were compared. The TCLP extract underestimated the amounts of oxoanion- forming elements leached, especially arsenic.

A different extraction procedure may be needed to test the hazard potential of waste because of the shortcomings of the TCLP. The State of California developed one alternative to the TCLP in the 1980's. This extraction method is called the Waste

Extraction Test (WET). The California WET is a more aggressive test than the TCLP because of a different extraction fluid, a longer mixing period, and a higher sample to extraction fluid ratio. Although the California WET method is similar to the TCLP, there are some important differences between the two tests as shown in Table 2.

California WET	TCLP
One set extraction fluid. Citric Acid Buffer pH 5.0	Extraction fluid selection depends on sample pH: a. Acetate buffer pH 4.93 b. Acetic Acid solution pH 2.88
Sample to extraction fluid ratio is 1:10	Sample to extraction fluid ratio is 1:20
Does not specify extraction vessel design	Requires extraction bottles made of glass or polypropylene
Requires use of 0.45 µm membrane filter for extract after extraction	Requires use of 0.6 to 0.8 µm glass fiber filter
Uses mechanical shaker for extraction	Requires rotary end over end fashion at 30 rpm
Extraction period of 48 hours	18 hours
Does not require acid digestion after extraction for metals	Requires acid digestion after extraction for metals other than mercury

**Table 1-2.** Comparison of TCLP and California WET, adapted from Sivia et al, 1991.

The California WET extraction fluid contains citric acid, which is a strong chelating agent. Because of this, the California WET usually extracts higher quantities of most metals than the TCLP test (Hooper *et al*, 1998). In previous studies, the California WET was found to extract two to ten times as much arsenic as the TCLP test, even after the TCLP value had been multiplied by two to account for the different dilution factors in the two tests (Hooper *et al*, 1998 and Sivia *et al*, 1991). California WET extract was also compared to actual municipal solid waste leachate in Hooper's study. The California WET extract generally extracted higher amounts of arsenic than was found in the municipal solid waste leachate. The overestimated level of arsenic extracted by the California WET was attributed to the existence of arsenic bound to iron as arsenopyrite in the waste. The citric acid chelates the iron and therefore solubilizes the arsenic.

Neither the TCLP test nor the California WET appear to be appropriate for determining the toxicity of arsenic. However, they are the tests that regulations are

currently based on. There is no indication of what the future extraction procedures will be, so until new extraction procedures are developed, the TCLP and the California WET will be used to characterize waste.

If waste fails the TCLP test (or the California WET test in California), it must be disposed of in a hazardous waste landfill. Hazardous waste must be closely monitored and hazardous landfills must be maintained. This disposal option is very expensive. Changes in the TCLP or California WET value may force increased residuals to be disposed in this manner.

### ***Storage and Dewatering of Arsenic Containing Residuals***

Handling procedures of arsenic-containing residuals will also face stricter regulation when a lower arsenic MCL is imposed. Handling procedures include transport, storage and dewatering of residuals. A practice of particular environmental concern is the storage of arsenic containing residuals in lagoons, which is a common practice for drinking water treatment residuals.

The long-term storage of arsenic containing residuals in lagoons may release arsenic that is present in the waste and cause soil or ground water contamination as well as dangerous working conditions. Currently, on-site lagoons are minimally regulated at the federal level under SDWA and RCRA (EPA, 2000b). The lagoon storage of arsenic laden residuals is a practice that is a potential risk, and may face stricter regulations in the future.

Although of great environmental and regulatory importance, little study has been done to determine the effects of lagoon storage on drinking water treatment plant residuals. However, there have been studies investigating the effects of lagoon storage on other types of arsenic containing waste, including sewage sludge and soil (Carbonell-Barrachina *et al*, 2000 and Masscheleyn *et al*, 1991). These studies may provide insight into the behavior of arsenic in drinking water treatment plant residuals.

Oxidation-reduction potential and pH play an important role in arsenic chemistry and are therefore important when investigating the fate of arsenic in arsenic containing wastes. The redox equilibrium between the two most commonly found forms of arsenic in water, As(III) and As(V), is highly pH dependent, as shown in the arsenic Eh-pH

diagrams in Figures 1 and 2. Arsenic in the +3 oxidation state is more soluble than arsenic in the +5 oxidation state; therefore, the species of arsenic determines the solubility. In addition, the increased solubility of some compounds in lower pHs, such as iron hydroxides, may lead to the release of iron-bound arsenic (Villa-Lojo *et al*, 1997).

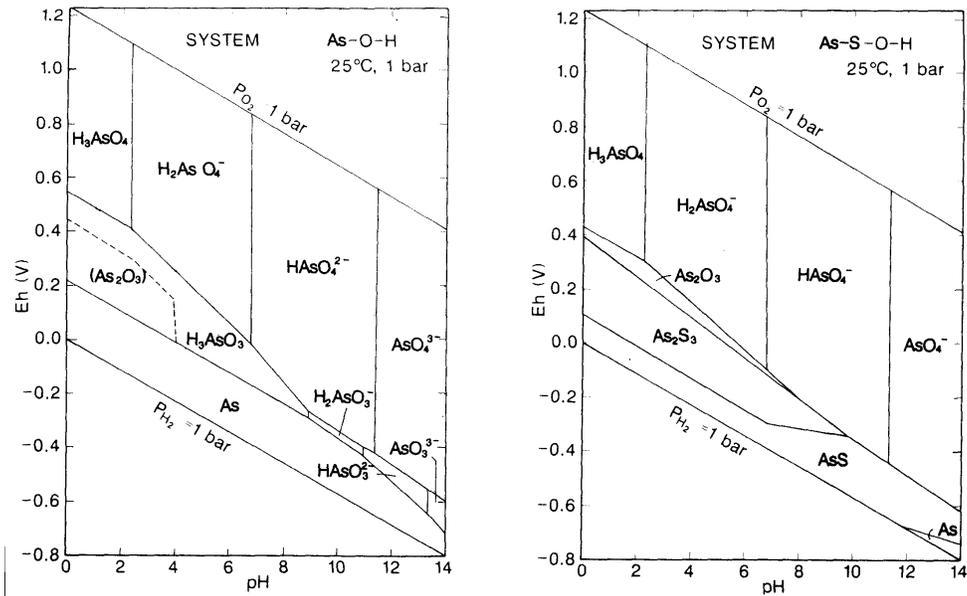


Figure 1-1. Eh - pH diagrams for the As-O-H and As-S-O-H systems, from Brookins, 1998.

Masschelyn *et al* (1991) studied the effect of pH and redox potential on arsenic solubility in contaminated soil and sediments. Soil and sediment samples were kept at redox potentials of -200, 0, 200 and 500 at varying pHs, and the soluble arsenic was measured for each sample. In general, more soluble arsenic was found in alkaline pH conditions. Changes in the redox potential had a large impact on the solubility of arsenic. As the redox potential decreased, arsenic solubility increased. The solubility changes were due to two mechanisms: the reduction of arsenate to arsenite and desorption and arsenic release due to the dissolution of iron-oxyhydroxides.

An experiment carried out by Carbonell-Barrachina *et al* (2000) studied the effects of redox potential and pH on arsenic solubility in sewage sludge. Separate samples of the sludge were adjusted to redox potentials -250, 0, 100, 325, and 450 mV with varying pHs. In general, arsenic was statistically more soluble in more neutral pHs than in acidic or basic pHs. Arsenic solubility was low under oxidizing conditions (325 and 400 mV). When reduced to 100 and 0 mV, arsenic solubility increased due to the

dissolution of iron oxyhydroxides. At very reduced conditions ( $-250$  mV), little soluble arsenic was present. Arsenic combined with sulfides to form arsenic sulfides, such as arsenopyrite ( $\text{AsFeS}$ ) or orpiment ( $\text{As}_2\text{S}_3$ ). The arsenic sulfides are insoluble and stable in reducing conditions; however, it was cautioned that sulfides are easily oxidized under aerobic conditions.

The redox potential and pH are characteristics that will affect all types of residuals. However, other factors are characteristic to the type of residuals present. The coagulant that is used can classify residuals. The main types of coagulants used are ferric chloride, aluminum hydroxide, and lime. The type of residual may affect the solubility of arsenic in water, and therefore the risks of lagoon storage of the residual.

The use of ferric chloride is now the most common means of arsenic removal from drinking water. Nevertheless, the long-term stability of the solid material produced is not known. The removal process has been proven to be mainly a result of adsorption and not precipitation of ferric arsenate (Robins *et al*, 1988 and 1991). This being so, the solid product is not suited for long-term ponding storage due to the risk of arsenic desorption and release (Twidwell *et al*, 1994).

No information is available about the effects of storage on residuals from alum coagulation plants. The soluble aluminum concentration in soil has been shown not to be redox sensitive (Masscheleyn *et al*, 1991). The mechanism of arsenic removal by aluminum hydroxide is through adsorption (Mok and Wai, 1994). As(III) does not sorb strongly to  $\text{Al}(\text{OH})_3$ . Consequently, the reduction of arsenate to arsenite and desorption of arsenic from aluminum hydroxide solids are possible.

The lagoon storage of arsenic laden softening residuals is dependent on environmental conditions. If the residuals are stored in an aerobic environment with an abundance of surface area exposed to the air, the arsenic will be released. One study verified that carbon dioxide in air converts the calcium arsenate to calcium carbonate and soluble arsenic (Robins, 1992). However, if the residuals are kept under reducing conditions, calcium arsenate formation will immobilize the arsenic (Bothe and Brown, 1999).

## ***Conclusions***

A new MCL for arsenic in drinking water is expected to be announced by the year 2002. With it will come stricter regulations on the disposal and handling procedures associated with arsenic containing residuals. Many unanswered questions remain about the arsenic content and toxicity of drinking water treatment plant residuals and the appropriate handling and disposal approaches. Different types of drinking water treatment plant residuals need to be characterized and subjected to toxicity tests. Additionally, the release of arsenic during handling procedures, including lagoon storage, needs to be explored. Without these important data, regulations may be implemented that are unrealistic and unattainable for drinking water treatment utilities.

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## METHODS

### Residual Acquisition

Seven drinking water treatment plants with high levels of arsenic in their raw water were contacted to participate in the research project. Included in the study are four plants that coagulate with ferric chloride, two with aluminum hydroxide, and one with lime.

Utility Name	Location	Coagulant Type
Billings	Billings, Montana	Ferric Chloride
City of Great Falls Water Treatment Plant	Great Falls, Montana	Aluminum Hydroxide
City of Helena Water Treatment Division	Helena, Montana	Aluminum Hydroxide
Department of Water and Power City of Los Angeles	Los Angeles, California	Ferric Chloride
Indiana America Water Company	Kokomo, Indiana	Ferric Chloride
Lockwood Water User Association	Billings, Montana	Ferric Chloride
Louisiana Water Company	New Iberia, Louisiana	Lime

**Table 2- 1.** Participating facilities, locations, and coagulant types.

Samplings kits containing pre-washed plastic cubitainers, sampling instructions, and pre-addressed boxes were mailed to the facilities. Two one-gallon containers of residuals were mailed back to Virginia Tech for analysis. Four five-gallon cubitainers of spent filter backwash water were sent to Environmental Engineering and Technology, Inc. for testing.

### Lagoon Simulations

As the two gallons of residuals were received from each utility, they were divided into four half-gallon borosilicate ground glass jars that had been acid washed in a 10% nitric acid bath. All of the lagoon simulations were conducted at the percent solids as received from the utility, with the exception of Billings. Billings residuals were received at 25.7% solids and were diluted with distilled water to 18.6% solids. The percent solids of the residuals in the lagoon simulations are given in Table 2.

<b>Residual</b>	<b>Percent Solids of Lagoon Simulation</b>
Billings	18.6
Great Falls	3.00
Helena	3.64
Indiana	1.86
Lockwood	4.73
Los Angeles	16.14
Louisiana	4.05

**Table 2- 2.** The percent solids of each residual in the lagoon simulations.

One jar of residuals from each facility was analyzed immediately. The remaining jars of residuals were sealed and stored in a 20°C controlled temperature room in darkness to simulate lagoon aging. The additional jars were analyzed every two months for six months.

To simulate different conditions that may exist in lagoons, lagoons at pH 4.8 and lagoons containing biodegradable organic matter (BOM) were created using extra residuals obtained from the Los Angeles utility. The lagoons at pH 4.8 simply had concentrated trace metal grade nitric acid added until they reached the desired pH. The biodegradable organic matter was added each BOM lagoon in the form of 10mls of anaerobic sewage sludge, 500 mg/L bactopectone, 500 mg/L glucose, and 200 mg/L sodium sulfate.

Every two months, another jar of residuals from each facility was sacrificed for analysis. The dissolved oxygen, oxidation-reduction potential, and pH were measured as indicated in Standard Methods. The residuals were separated into liquid and solid components by vacuum filtration through a Büchner funnel with a Whatman 41 filter (20-25 µm pore size).

The liquid portions of the residuals were acid digested using EPA Method 3010a. The digestates were filtered through 0.45 µm pore size filter syringe filters and analyzed with ICP-AES using EPA Method 6010. A Concomitant Metals Analyzer (CMA)

hydride generator using sodium tetraborohydride and hydrochloric acid was used to increase the arsenic detection limit and reduce possible interferences.

The solids portions after filtration were at approximately 20% solids; the actual solid concentrations are given in Table 3. After two months and six months of storage, the cake solids were tested for toxicity using EPA Method 1311: Toxicity Characteristic Leachate Procedure (TCLP). HDPE containers that had been acid washed were used as the extraction vessels. The TCLP extract was acid digested using EPA Method 3010a and analyzed using Method 6010b with CMA

<b>Residual</b>	<b>Percent Solids After Filtration</b>
Billings	30.4
Great Falls	17.2
Helena	4.1
Indiana	12.6
Lockwood	22.6
Los Angeles	20.0
Louisiana	31.4

**Table 2- 3. The percent solids of each residual after filtration.**

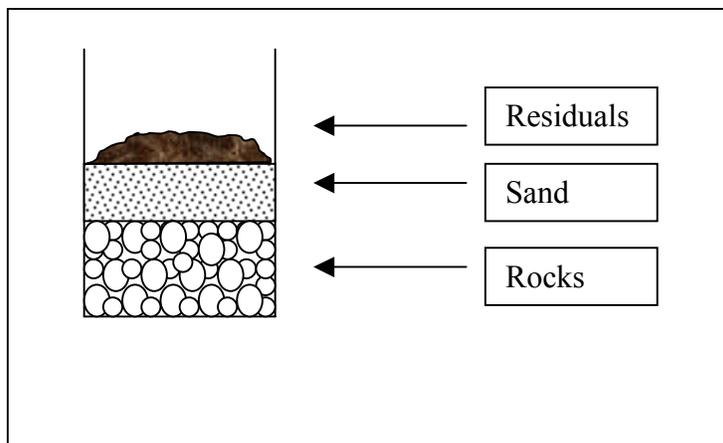
The California Hazardous Material Laboratory SOP 910: Waste Extraction Test (WET) was also performed on the residuals' filtered solids. Acid washed HDPE bottles were used as the extraction vessels. The California WET extract was diluted by a factor of ten and analyzed with EPA Method 6010 with CMA.

The remaining solid portions of the residuals were dried for 48 hours in a 100° C oven and then acid digested using EPA Method 3050b. The digestate was filtered through a 0.45 µm pore size syringe filter and diluted with acidified nanopure water by a factor of ten. It was then analyzed using EPA Method 6010 with CMA.

The dry solid portions of the residuals were also subjected to the TCLP and California WET following the same procedures as at approximately 20% solids. The extracts were analyzed with ICP-AES with CMA.

## Bench Scale Sand Drying Beds

Five Bench-Scale Sand Drying Beds were constructed using six-inch diameter PVC pipe with sealed bottoms. The sand drying beds were filled with rocks, then sand. A sampling tube at the bottom of the sand bed allowed for sampling of the leachate. Figure 1 is a representation of a cross section of the constructed sand drying beds.



**Figure 2- 1.** A cross sectional diagram of a bench-scale sand drying bed.

In four of the sand beds, approximately 700 g of Los Angeles residuals were placed on top of the sand and allowed to dry. One sand bed contained no residuals and served as the control. All of the sand beds were placed outside for exposure to natural environmental conditions and sunlight. Once a week, the sand beds were watered with approximately an inch of simulated rain. The leachate was collected, acid digested using EPA Method 3010a and analyzed using Method 6010 with CMA.

The four sand beds containing residuals were subject to differing conditions. One sand bed was watered with simulated rain at pH 5.8. Another was also watered with pH 5.8 rain, but covered so as not to allow exposure to UV light. Two were watered with simulated acid rain, one at pH 4.8 and another at pH 3.8.

The simulated rain was created by adding a small amount of ionic strength to distilled water in the form of sodium chloride and adjusting the pH by purging with gas. To increase the pH, nitrogen was used. To decrease the pH, carbon dioxide was used. Acid rain was created by adding 1 N trace metal grade nitric acid until the desired pH was achieved.

# THE EFFECTS OF LAGOON STORAGE AND SAND BED DRYING ON RESIDUALS CONTAINING ARSENIC

CORTNEY H. ITLE, JOHN T. NOVAK, MARC EDWARDS

## INTRODUCTION

### *Background*

With a new arsenic MCL for drinking water looming on the horizon, drinking water treatment utilities may need to comply with new arsenic regulations. Stricter regulations are likely to be placed on the handling and disposal procedures of arsenic containing residuals. Handling procedures for residuals include storage, dewatering and transport practices. Currently, several ultimate disposal options available for the disposal of arsenic containing wastes are available, including discharge into sewers, land application, and landfilling. Changes in regulation can impact each of these handling and disposal options.

Although of great environmental and regulatory importance, little study has been done to characterize the fate of arsenic in drinking water treatment residuals or to determine the effects of different handling procedures on arsenic mobility. The concentration of arsenic in residuals governs the disposal methods that may be used. Residuals may also contain arsenic that may leach and contaminate water that is discharged to surface or groundwater. The potential of arsenic to be released from the residuals during lagoon storage or dewatering is a concern that must be evaluated and factored into dewatering operations.

Land disposal of arsenic containing residuals is governed by the concentration of arsenic in the solids, as regulated under 40CFR Part 503<sup>1</sup>. Residuals containing less than 41 mg/kg arsenic can be applied to land with no limitations, while residuals containing 75 mg/kg arsenic or less can be applied in limited quantities. Land disposal is defined as the placement of residuals into specified sludge landfills called monofills. The limit upper for land disposal is 73 mg/kg arsenic.

Another option available for the disposal of residuals is their placement into municipal solids waste landfills. Only non-hazardous waste can be disposed in this manner. The hazardous characteristic of concern for arsenic containing residuals is the toxicity. Currently, under the Resource Conservation and Recovery Act (RCRA), the Toxicity Characteristic Leachate Procedure (TCLP) is the nationally authorized toxicity test. The TCLP has some flaws, including that it may under-predict the amount of arsenic that leaches from waste and it does not take into account the affect of changes in the redox potential<sup>2,3</sup>. The state of California developed its own toxicity test called the California Waste Extraction Test (WET). The California WET is a chemically more aggressive test, but has some of the same limitations as the TCLP test.

Certain handling procedures are also currently regulated. On-site lagoons are minimally regulated at the federal level under Safe Drinking Water Act (SDWA) and RCRA<sup>4</sup>. Stricter regulations are expected with the possible

implementation of new arsenic regulations. Arsenic residuals stored in lagoons may release arsenic to solution and contaminate surface or groundwater. There have been no studies on the impact of lagoon storage of residuals on arsenic mobility. However, there have been studies investigating the effects of lagoon storage on other types of arsenic containing waste, including sewage sludge and soil<sup>5,6</sup>. These studies may provide insight into the behavior of arsenic in drinking water treatment plant residuals.

No controls are presently in place for dewatering practices, and no studies have investigated the potential for arsenic to become mobilized during these practices. The drying of residuals on sand drying beds has the potential to cause increased environmental exposure to arsenic. The sand drying beds are usually placed in the ground and exposed to the elements. Rainwater may cause arsenic to leach from the residuals into surface or groundwater. The pH of the rainfall might be an important consideration. Exposure to sunlight might also play a role in arsenic release.

### ***Chemistry of Arsenic***

Oxidation-reduction potential and pH play important roles in arsenic speciation and solubility. Therefore, both of these are important parameters to consider when determining the fate of arsenic in a system. The redox equilibrium between the two most commonly found forms of arsenic in water, As(III) and As(V), is highly pH dependent. Arsenic in the +3 oxidation state is more soluble than arsenic in the +5 oxidation state. In addition, the increased solubility of some arsenic binding compounds in lower pHs (such as iron hydroxides) may lead to the release of iron-bound arsenic<sup>7</sup>.

Previous studies on sewage sludge and soil found soluble arsenic to increase under reduced conditions<sup>5,6</sup>. However, at very reduced conditions (–250 mV), little soluble arsenic was present. The increased arsenic solubility under reduced conditions was due to two mechanisms, the reduction of arsenate to arsenite and desorption, and arsenic release due to the dissolution of iron-oxyhydroxides. Under very reduced conditions, arsenic combined with sulfides to form compounds, such as realgar (AsS), orpiment (As<sub>2</sub>S<sub>3</sub>), or arsenopyrite (AsFeS). Arsenic sulfides are insoluble and stable under reducing conditions.

### ***Residuals***

Ferric chloride is now the most commonly used coagulant for arsenic removal from drinking water. Nevertheless, the long-term stability of the arsenic/ironhydroxides material produced is not known. The removal process has been proven to be mainly a result of adsorption to ferric oxyhydroxides and not by the precipitation of ferric arsenate<sup>8</sup>. This being so, the solid product is not suited for long-term ponding storage due to the potential for arsenic desorption and release<sup>9</sup>.

There is no information available about the effects of storage on residuals from alum coagulation plants. The soluble aluminum concentration in soil has been shown to be redox insensitive<sup>6</sup>. The mechanism of arsenic removal by aluminum hydroxide is through adsorption<sup>10</sup>. Consequently, the reduction of arsenate to

arsenite under reduced conditions and subsequent desorption of arsenic from aluminum hydroxide solids is possible.

The lagoon storage of arsenic laden softening waste is very dependent on environmental conditions. If the waste is stored in an aerobic environment with an abundance of surface area exposed to the air, arsenic may be released. One study showed that carbon dioxide in air converts the calcium arsenate to calcium carbonate and soluble arsenic<sup>8</sup>. However, if the waste is kept under reducing conditions, calcium arsenate formation should immobilize the arsenic<sup>11</sup>. This is the opposite of that expected for stored ferric hydroxide residuals.

### ***Objectives***

There is little data available quantifying the amounts of arsenic present in drinking water treatment residuals, or the potential of arsenic to leach out of residuals during lagoon storage and sand drying. The amounts of arsenic present determine the disposal options of residuals. Few regulations exist for lagoon storage or sand bed drying, but the potential for arsenic to leach out may present risks that should be addressed in future regulations.

This study is designed to increase our knowledge about the fate of arsenic in arsenic containing residuals. The concentrations of arsenic present will be measured to establish a range for a variety of arsenic-containing residuals and this will define which disposal options are available for arsenic containing residuals. Lagoon storage will be simulated using residuals from utilities that treat with ferric chloride, aluminum hydroxide, and lime (softening plant) to determine if, under what conditions, and to what extent arsenic is released. The residuals will undergo toxicity testing to determine the potential for disposal in municipal solid waste landfills. Bench scale sand drying beds will be used to test the potential of arsenic to leach from residuals on sand beds.

## **METHODS**

Seven drinking water treatment plants with quantifiable arsenic concentrations in their raw water were contacted to participate in the study. Sampling kits containing pre-washed plastic cubitainers and sampling instructions were sent to the utilities. Of the seven plants, four used ferric chloride as their coagulant, two used aluminum hydroxide, and one was a softening plant using lime. Eight liters of residuals from each utility were sent back to Virginia Tech to undergo analysis.

The residuals were chemically characterized to determine the concentrations of arsenic present. Lagoon simulations were created and analyzed over time for soluble arsenic. Toxicity testing was performed on the lagoon residuals. Bench scale sand drying beds were constructed and the leachate was analyzed for arsenic.

### ***Lagoon Simulations***

As the residuals from each utility were received, they were divided into four 2-liter acid washed glass jars. The first jar of residuals from each plant underwent analysis immediately. The remaining jars were sealed and stored in darkness at

20°C to simulate lagoon storage. The contents of the stored jars were analyzed over a period of six months.

Lagoons with added biodegradable organic matter were also created. Anaerobic sewage sludge, bactopeptone, glucose, and sodium sulfate were added to jars containing extraneous Los Angeles residuals. These simulated lagoons were stored and sampled similar to the others.

Every two months, another jar of residuals from each source was sacrificed and the dissolved oxygen, redox potential, and pH were measured and recorded. The residuals were separated into solid and liquid portions using a Whatman #41 filter (20-25 µm pore size). The liquid portions of the residuals were acid digested and analyzed with ICP-AES (EPA Methods 3010a and 6010) for total iron, aluminum, and arsenic. A hydride generator was used to increase the arsenic detection limit of the instrument.

During the second and sixth month, the solid portions of the filtered residuals were subjected to toxicity testing. The TCLP (EPA Method 1311) and California Waste Extraction Test (WET, California HML SOP 910) were performed. The TCLP extract was acid digested with EPA Method 3010a. The TCLP digestate and California WET extract were analyzed with ICP-AES with a hydride generator.

The remaining solid portions of the residuals were oven dried for 48 hours in a 100°C oven. The dried residual solids were then acid digested (EPA Method 3050b) and analyzed for total iron, aluminum, and arsenic using ICP-AES.

### ***Sand Drying Beds***

Five bench scale sand drying beds were constructed using 6-inch diameter PVC pipe that had been sealed on the bottom and filled with rocks, then sand. A sampling tube was placed at the bottom of the column to allow for collection of the leachate. One drying bed had no sand added and served as the control. Four of the sand drying beds had residuals placed on top of the sand.

The residuals used in the sand drying beds were from Los Angeles. The Los Angeles utility sent supplementary residuals to undergo additional testing. Los Angeles is a utility that coagulates with ferric chloride and contains high amounts of arsenic

The four drying beds containing residuals were subjected to differing conditions. One sandbed was watered with simulated rain at pH 5.8 and exposed to sunlight. Another was watered with pH 5.8 rain, but covered so as not to allow exposure to sunlight. Two of the units were watered with simulated acid rain, one at pH 4.8 and another at pH 3.8 and exposed to sunlight.

Simulated rainfall was created by adding a small amount of sodium chloride to distilled water and adjusting the pH by purging with gas. To increase the pH, nitrogen was used; to decrease the pH, carbon dioxide was used. Acid rain was created by adding 1 N trace metal grade nitric acid until the desired pH was achieved.

## RESULTS AND DISCUSSION

Table 3-1 lists the total arsenic, aluminum, and iron concentrations for each residual, as well as the type of coagulant used and the arsenic to coagulant molar ratio. All of the residuals contain significant amounts of arsenic, considering that current regulations consider “clean” residuals to contain 41 mg/kg arsenic or less (Edwards, 2000). According to the same regulations, residuals with up to 75 mg/kg arsenic can be land applied in limited amounts. Using these regulations, only the Billings residuals could be land applied.

The Indiana residuals contain an extremely high amount of arsenic. However, the Indiana plant uses ferric chloride as a coagulant and it also contains a high fraction of iron. The arsenic to iron molar ratios are in the same range as the other ferric chloride utilities in the study.

	Percent Solids	As mg/kg*	Fe g/kg*	Al g/kg*	Coagulant	As:Fe molar ratio	As:Al molar ratio
<b>Billings, MT</b>	18.6	72.2	74.7	25.5	Ferric Chloride	0.001	-
<b>Great Falls, MT</b>	3.00	490.2	14.7	109.2	Aluminum Hydroxide	-	0.002
<b>Helena, MT</b>	3.64	421.9	43.7	41.0	Aluminum Hydroxide	-	0.004
<b>Indiana, IN</b>	1.86	15729.6	4932.2	1.7	Ferric Chloride	0.002	-
<b>Lockwood, MT</b>	4.73	817.7	327.7	11.8	Ferric Chloride	0.002	-
<b>Los Angeles, CA</b>	16.14	726.1	139.4	17.0	Ferric Chloride	0.004	-
<b>Louisiana, LA</b>	4.05	109.4	2.3	28.3	Lime	-	-

**Table 3- 1.** Physical and chemical characteristics of each residual, including the percent solids, the total arsenic, iron, and aluminum concentrations, the coagulant used, and the molar ratios of arsenic to the coagulant metal.

\*Percent solids are expressed on a dry basis.

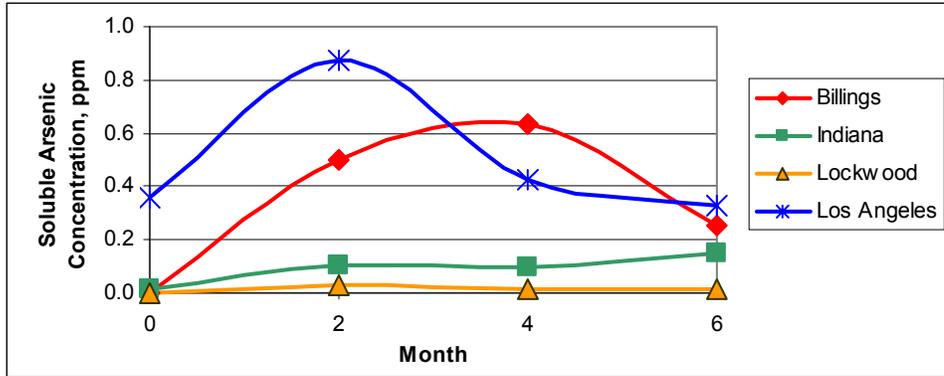
### Lagoon Simulations

Lagoon storage affected each residual differently, depending on physical and chemical characteristics of the residual. At the end of the six-month storage period, the lagoons ranged from no increase in soluble arsenic (Louisiana) to a 125-fold increase in soluble arsenic (Billings). Several of the residuals saw increases and subsequent decreases during storage (Billings and Los Angeles). Others continued to increase throughout the storage period (Indiana). Because of the many reactions that may be occurring and the different characteristics of each residual, it is difficult to determine exactly the mechanisms that result in arsenic release. Several mechanisms may be at work for any one residual.

### *Ferric Chloride Residuals*

Figure 3-1 shows the soluble arsenic concentration for each ferric chloride residual over the six-month storage period. Residuals from Los Angeles and Billings had considerably higher percent solids than the other residuals (see Table 3-1). The higher percent solids would indicate the presence of more arsenic, which might account for Los Angeles and Billings residuals’ high soluble arsenic

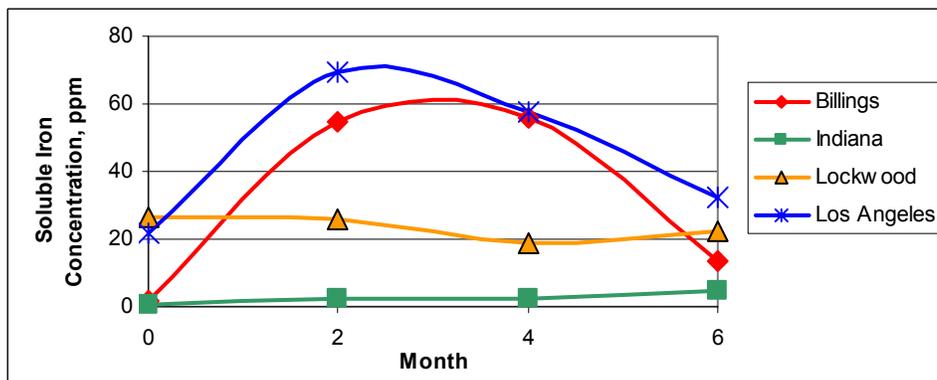
concentrations. Additionally, the higher percent solids result in a higher concentration of organic matter and this would cause a lower redox potential. The presence of additional organic matter could produce a decrease in the redox potential, which would cause an increase in soluble arsenic.



**Figure 3-1.** Soluble arsenic concentrations over six months in the ferric chloride residuals

The Indiana residuals, which contained high concentrations of arsenic in the solids, did not release much arsenic to solution. This can be explained by two factors. First, the Indiana residuals were very dilute, at only 1.86% solids. In addition, the Indiana residuals had a high redox potential, around 200 to 300 mV, during the entire six months of storage. At high redox potentials, little soluble arsenic is present. As mentioned earlier, the solid concentration affects the concentration of organic matter. The high redox potential in the Indiana residuals could be caused by a low concentration of organic matter as a consequence of the low percent solids.

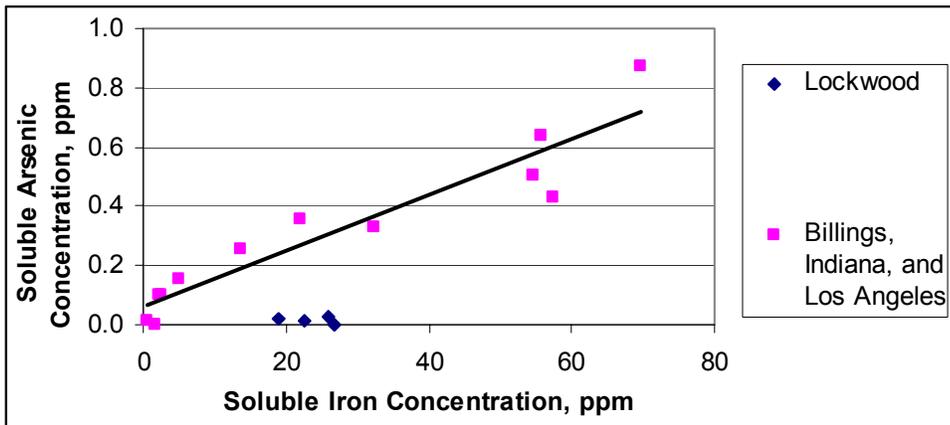
Figure 3-2 presents the soluble iron present in the ferric chloride lagoons over the six month storage period. When comparing the soluble arsenic concentrations (Figure 1) to the soluble iron concentration (Figure 2), the relationship is apparent. Los Angeles and Billings residuals contain the highest concentration of both soluble arsenic and soluble iron. The soluble arsenic and iron concentrations both decrease in Los Angeles and Billings residuals and both remain nearly unchanged in the Indiana and Lockwood residuals.



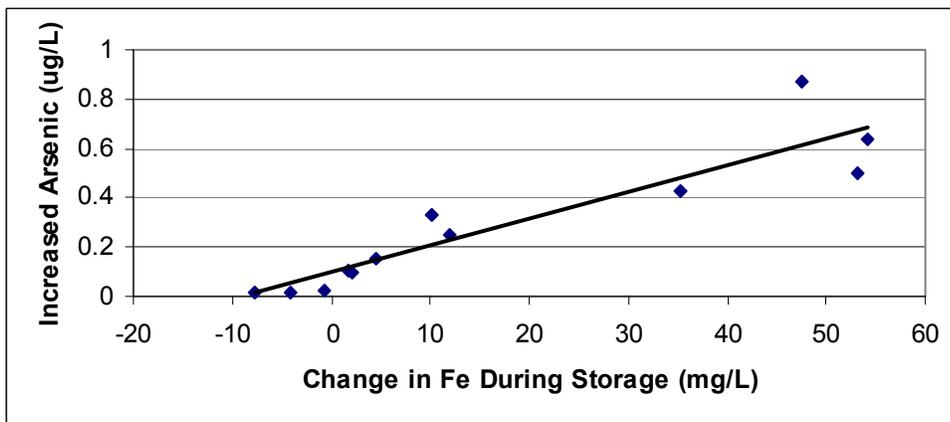
**Figure 3-2.** Soluble iron concentration in ferric chloride residuals over six months.

Figure 3-3 is plot of the soluble arsenic and iron concentrations in the ferric chloride residual lagoons. The arsenic and iron concentrations are highly correlated for all of the ferric chloride residuals except Lockwood. The Lockwood residuals had unusually low arsenic concentrations for the concentration of iron present. Lockwood also had a high initial concentration of iron in solution, and the value did not change significantly over time. The  $R^2$  value of the best-fit line for Billings, Indiana, and Los Angeles is 0.89.

Figure 3-4 is a plot of the soluble arsenic concentrations versus the change in soluble iron concentrations. The change in soluble iron is the difference between the iron concentration at time zero and the value at each of the later storage times. This graph better presents what is occurring in the lagoons. The  $R^2$  value of this plot is 0.85. The soluble arsenic concentrations are affected by the change in soluble iron and not necessarily the concentration of soluble iron. As more iron is released to solution, more arsenic is also released.



**Figure 3-3.** Soluble arsenic vs. soluble iron in the ferric chloride residuals over six months of storage.



**Figure 3-4.** Soluble arsenic concentration versus the change in iron concentration in the ferric chloride residuals over six months of lagoon storage.

### Aluminum Hydroxide Residuals

The soluble arsenic concentrations in the lagoons containing aluminum hydroxide residuals are shown in Figure 3-5. Great Falls residuals contained a high concentration of soluble arsenic, which increased during the second month of storage, then decreased. The Helena residuals showed a slight increase in soluble arsenic over the six months of storage.

Figure 3-6 shows the soluble iron and aluminum concentrations in the Great Falls and Helena residuals over the storage period. The soluble aluminum concentrations in the lagoon residuals did not follow any patterns. Although the aluminum concentrations did fluctuate in the residuals over time, the changes were not related to the redox potential or the release of arsenic.

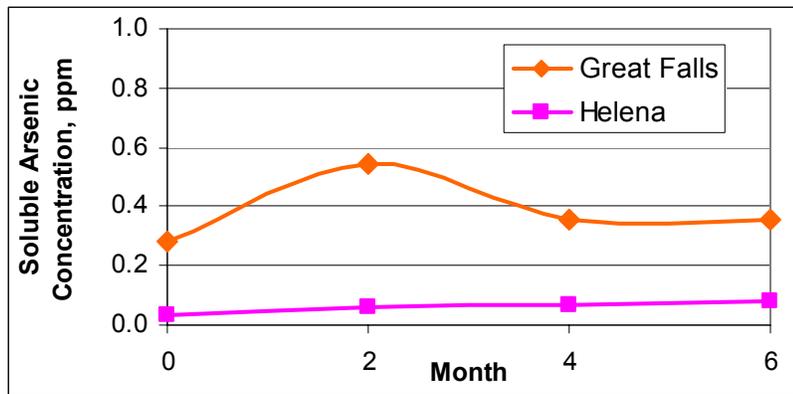


Figure 3- 5. The soluble arsenic concentration in aluminum hydroxide residuals over six months.

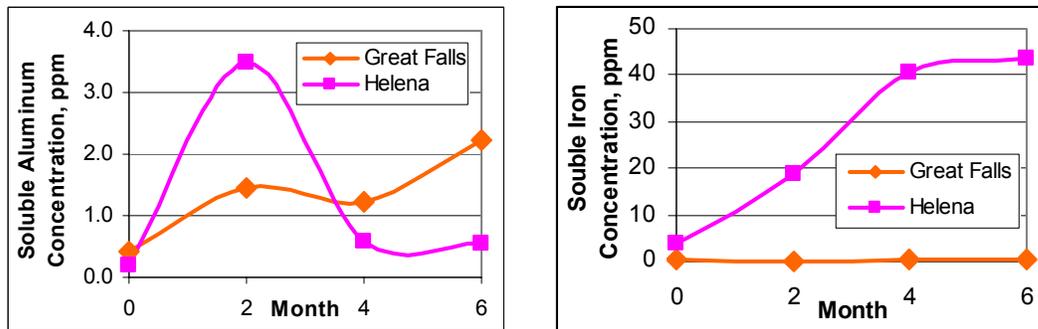


Figure 3- 6. The soluble aluminum and iron concentrations over six months in stored aluminum hydroxide residuals.

Great Falls and Helena are both alum residuals that have similar characteristics, including pH, organic content, percent solids, and arsenic concentrations in the dry solids. However, Great Falls has a much higher soluble arsenic concentration than Helena. The release of arsenic by Great Falls residuals might be related to a characteristic that was not identified in this study. Arsenic binds to clay, so a high concentration of clay in the Helena residuals would explain

the presence of less soluble arsenic. Additionally, it was found throughout the course of this study that the presence of multivalent cations, such as calcium and magnesium, increase the effectiveness of iron and aluminum at removing arsenic. A high calcium content in the Helena residuals would also explain the absence of soluble arsenic.

Another explanation of the difference in soluble arsenic concentrations in the aluminum hydroxide residuals might be presence of iron. The only difference between the two residuals detected in this study was that Helena contains a higher solid iron concentration than Great Falls. It could be speculated that aluminum hydroxide is not a stable solid. If arsenic easily desorbs from aluminum hydroxide solids, arsenic may bind to iron that is present. Because Great Falls residuals have little iron, the arsenic would be released to solution. The high soluble arsenic concentration in the Great Falls residuals may be due to the low iron concentration.

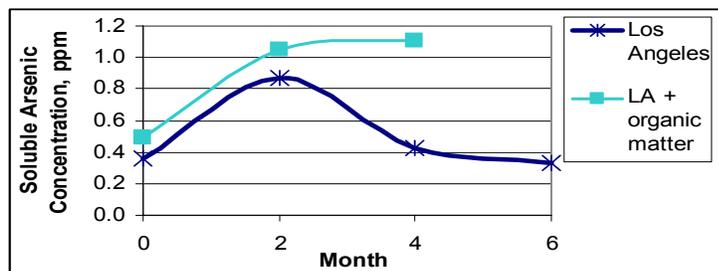
### ***Softening Residuals***

The Louisiana softening sludge was dilute (4.05% solids) and contained only 109.9 mg/kg arsenic. The amount of chemicals added to treat and soften water may act to dilute the arsenic concentration of the residuals. The softening residuals had the least amount of arsenic release, starting at zero, increasing to a maximum of only 5.4 µg/L arsenic, and then decreasing to zero. The softening sludge maintained a high redox potential, so there was no release of arsenic due to reduced conditions. The lagoons were sealed and not exposed to air, so even at high redox potentials there was no release of arsenic as a result of exposure to carbon dioxide. Calcium arsenate formation immobilized arsenic as predicted by Bothe and Brown (1999)<sup>11</sup>.

### ***Biodegradable Organic Matter***

To determine the effects of biodegradable organic matter on arsenic solubility, lagoons containing Los Angeles residuals with added organic matter were maintained. The soluble arsenic concentrations of the Los Angeles residuals and the Los Angeles residuals with added organic matter are presented in Figure 3-7. From time zero, the residuals with organic matter release more arsenic to solution.

Arsenic is similar to phosphorus, and it can be used by organic matter in the same manner as phosphorus. Bacteria consume the arsenic and convert it to organic forms, which are more soluble. The presence of organic matter increases arsenic solubility.

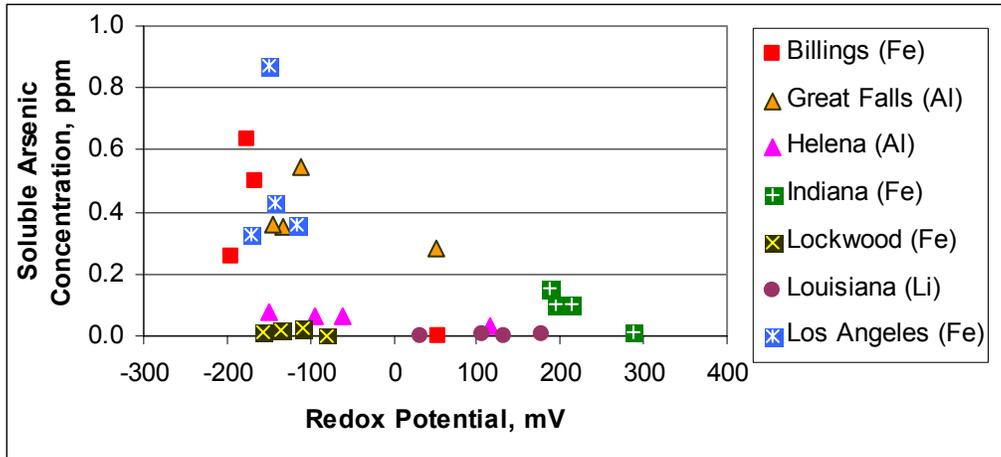


**Figure 3- 7.** Soluble arsenic in Los Angeles residuals and Los Angeles residuals with added organic matter.

### Redox Potential

Each residual had a different response to storage. The varied responses can be partially explained by the redox potentials measured in each lagoon over time. Figure 3-8 shows a plot of the soluble arsenic concentration vs. redox potential for all of the residuals. In general, as the redox potential decreases, the soluble arsenic concentration increases. Previous studies have also documented this trend and shown that this result from the reduction of arsenite to arsenate and desorption, and the reduction of iron hydroxides and the subsequent release of arsenic<sup>6</sup>.

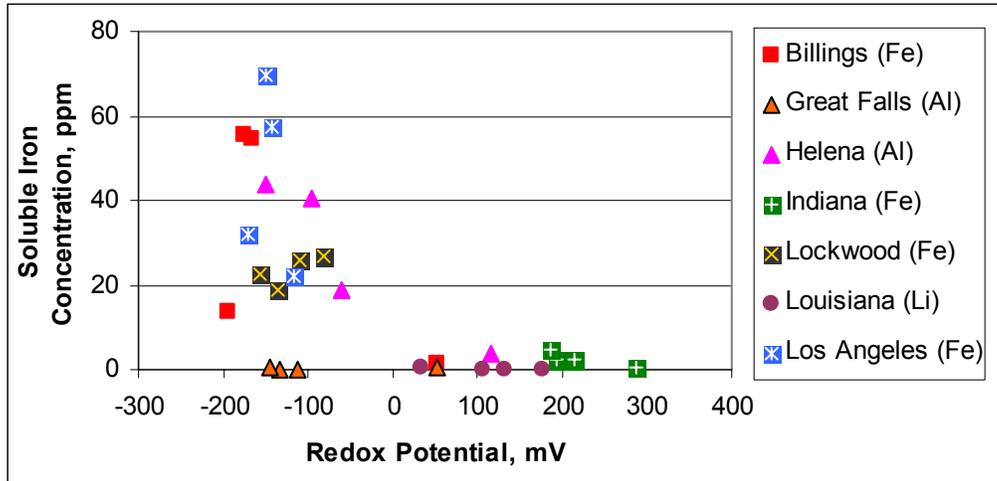
During the sixth month of storage, the soluble arsenic in the Billings and Los Angeles lagoons decreased, although the redox potentials remained very low (-195 and -171 mV). Previous studies recorded that the soluble arsenic concentrations start to decrease under very reduced conditions (-250mV), most likely due to the formation of arsenic sulfides, which are insoluble in reducing conditions<sup>5</sup>. The formation of arsenic sulfides might be one explanation for the decrease in soluble arsenic of some of the lagoons. However, this response is more common in conditions more reduced than the lagoons. Additionally, previous studies noting this behavior saw a decline in the relationship between arsenic and iron<sup>5</sup>. As demonstrated earlier, there is a strong association between arsenic and iron release. The decreases in soluble arsenic in Billings and Los Angeles residuals are most likely related to the decreases in soluble iron and not due to the formation of arsenic sulfides.



**Figure 3- 8.** The soluble arsenic concentrations in the lagoon residuals vs. the redox potentials.

Ferric chloride residuals are represented with squares, aluminum hydroxide residuals with triangles, and the softening residuals with a diamond.

As would be expected due to the high correlation between arsenic and iron, soluble iron also increases with decreasing redox potential, as shown in Figure 3-9. However, also similar to arsenic, soluble iron decreases in Billings and Los Angeles residuals even though the redox potentials decrease.

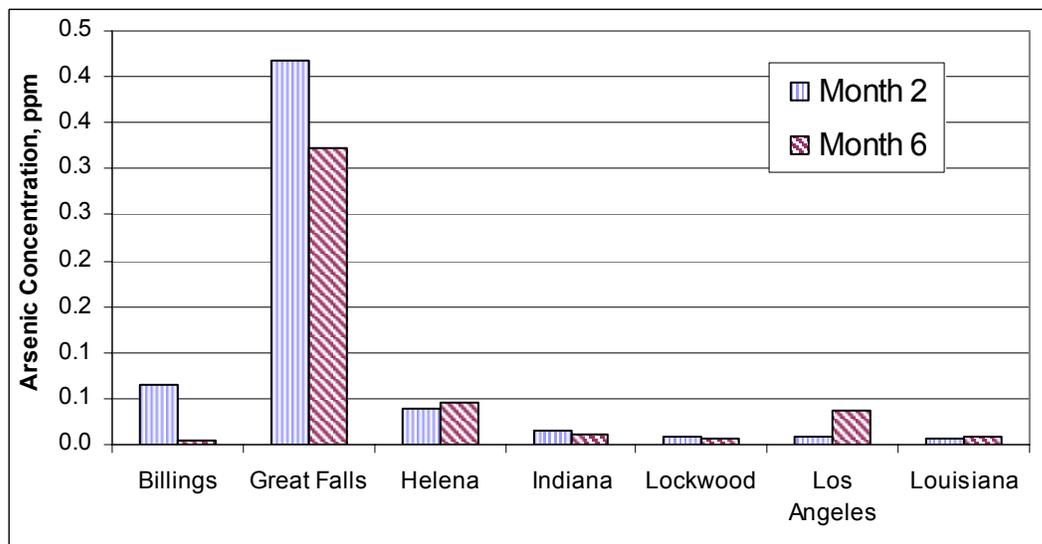


**Figure 3- 9.** Soluble iron in residual lagoons vs. redox potentials.

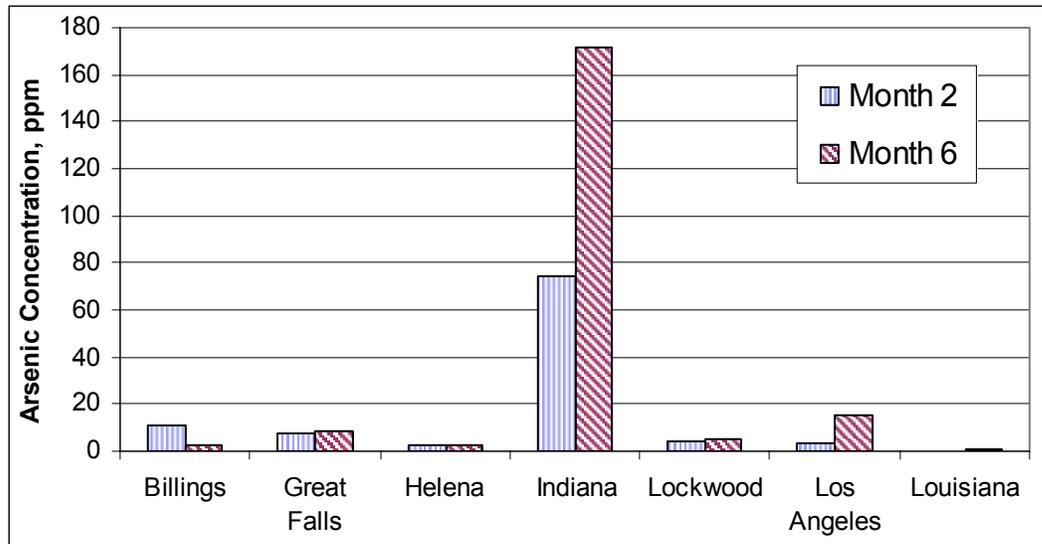
Ferric chloride residuals are represented with squares, aluminum hydroxide residuals with triangles, and the softening residuals with a diamond.

### *Effect of Storage on Toxicity Testing*

California WET and TCLP tests were performed on the sludge at approximately 20% solids during the second month and again during the sixth month, as shown in Figures 3-10 and 3-11. The California WET is a more aggressive extraction procedure and it extracted more arsenic than the TCLP, similar to previous studies<sup>2</sup>. The regulatory limit for both tests is currently 5 mg/L arsenic, concentrations greater than 5 mg/L value label the sludge as hazardous. All of the residuals pass the TCLP test, but only three of the residuals pass the California WET test at months 2 and 6.



**Figure 3- 10.** TCLP arsenic concentrations of the residuals after 2 months of storage and 6 months of storage.



**Figure 3- 11.** California WET arsenic concentrations of the residuals after two months of storage and after 6 months of storage.

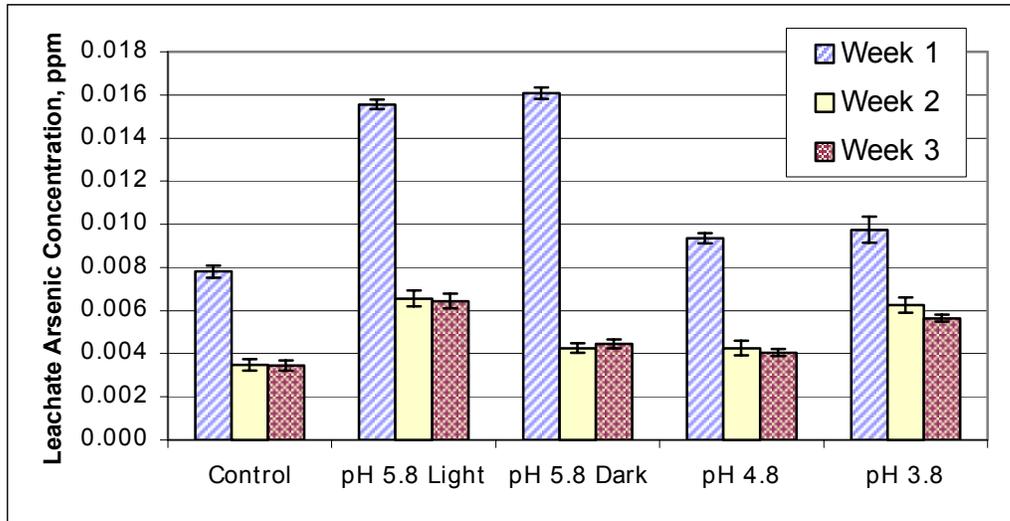
The changes in soluble arsenic concentration over time affect the TCLP and California WET results. Billings provides a good example of the effects of soluble arsenic on the TCLP and California WET results. At two months, Billings contained one of the highest concentrations of soluble arsenic. By six months, the arsenic concentration had decreased greatly. Looking at Figures 3-10 and 3-11, the TCLP and California WET values also decrease over time. The Billings residuals did not pass the California WET at month 2, but did pass at month 6.

Indiana is another example of the effects of soluble arsenic on toxicity testing. The soluble arsenic concentration in Indiana residuals increased over time. The TCLP value remained nearly the same, but the California WET increased dramatically.

The exception to the soluble arsenic concentrations affecting toxicity testing is the Los Angeles residuals. Although the soluble arsenic concentrations decrease from two months to six months, the TCLP and California WET values increase. The Los Angeles residuals pass the California WET test at month 2, but do not pass at month 6.

### Sand Drying Beds

Figure 3-12 shows the results of the bench-scale sand drying bed testing on the Los Angeles residuals. The arsenic concentration in the leachate from each sand bed is shown over the three week testing period. The arsenic concentrations in the leachates were also measured at four and five weeks and found to be below the detection limit of the ICP-AES (2 µg/L arsenic). Although the Los Angeles residuals contain 726.1 mg/kg arsenic, very little arsenic leached from the sand drying beds. After subtracting the control leachate from the experimental leachates, all of the arsenic concentrations are below 10 ppb.



**Figure 3- 12.** The arsenic concentrations of sandbed leachates over a three week period.

The outside layers of the residuals dry first, creating an impermeable layer. As the sand drying beds are watered, only the outside layers of the residuals are rinsed. Because the outside layers are dry, there is no soluble arsenic present and the water is neither acidic nor reduced enough to solubilize any arsenic.

## CONCLUSIONS

Drinking water treatment plants that have significant amounts of arsenic in their raw water produce residuals that contain high concentrations of arsenic. In accordance with current regulations, the majority of these residuals will not be land applicable. Other disposal options will have to be utilized.

The lagoon storage of arsenic containing residuals has different effects on arsenic release. Residuals stored at higher percent solids exhibit higher soluble arsenic concentrations. The release of arsenic in ferric chloride residuals is directly related to the change in soluble iron. The release of arsenic in aluminum hydroxide residuals might be related to the amount of iron present, but other, unknown factors, appear to be important. Softening residuals released less than 6 ppb arsenic, due to high redox potentials and a low concentration of arsenic.

The presence of biodegradable organic matter increases the arsenic solubility. Residuals containing high concentrations of organic matter should be more closely monitored for arsenic release.

As redox potentials decrease, soluble arsenic increases. Soluble iron also increases with decreasing redox potentials. The storage of residuals in lagoons creates reduced environments where arsenic will be released, possibly to surface and groundwater. Lagoon storage of arsenic containing wastes should be more strictly regulated in the future to prevent contamination.

TCLP and California WET results are affected by lagoon storage of the residuals. As residuals age, changes in the redox potentials influence changes in the soluble arsenic concentrations. The amount of soluble arsenic present influences

the results of the toxicity tests. Neither of the toxicity tests takes into account the changes that occur in redox potential, which is a serious failure for wastes containing arsenic.

The leachate of residuals on bench-scale sand drying beds contained less than 10 ppb arsenic. As long as the drinking water MCL is not set below 10 ppb arsenic, the leachate from sand beds is not of concern. The residuals air dry on the sand beds under oxidized conditions, so little soluble arsenic is present to leach off the residuals.

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# **DISPOSAL ALTERNATIVES OF DRINKING WATER TREATMENT RESIDUALS CONTAINING ARSENIC**

CORTNEY H. ITLE, JOHN T. NOVAK, MARC EDWARDS

## **INTRODUCTION**

A new, lower arsenic maximum contaminant level (MCL) for drinking water is expected to be announced within the year. As a result, stricter regulations are likely to be placed on the disposal of arsenic containing residuals. There are several disposal alternatives available for drinking water treatment residuals. The waste can be combined with sewage sludge and land applied to help fertilize soil. The residual waste can also be disposed in sludge specific landfills, called monofills. Non-hazardous residuals can be landfilled in municipal solid waste landfills. Residuals that are characterized as hazardous must be disposed of in hazardous waste landfills.

Land application is limited by the amount of arsenic present in the residuals<sup>1</sup>. Residuals containing less than 41 mg/kg arsenic can be applied to land with no limitations, while residuals containing 75 mg/kg arsenic or less can be applied in limited quantities. The limit for land disposal in monofills is 73 mg/kg arsenic.

To be disposed in municipal solid waste landfills, waste must be characterized as non-hazardous. For arsenic containing wastes, the main hazardous characteristic of concern is the toxicity. Presently, the toxic potential of waste is nationally characterized by the Toxicity Characteristic Leachate Procedure (TCLP), which was implemented in 1986 as part of the Resource Conservation and Recovery Act (RCRA). The purpose of the TCLP test is to simulate landfill conditions. The intent is to determine if the waste can be safely landfilled without fear of contaminants leaching from the waste and polluting the soil or groundwater. In the TCLP test, waste is combined with an acetic acid solution and shaken for 18 hours. The resulting TCLP extract, or liquid phase, is analyzed for the contaminant. For arsenic, the TCLP extract limit is 5 mg/L. If the TCLP extract contains less than 5 mg/L arsenic, the waste is considered non-hazardous. The TCLP value of 5 mg/L is derived from the current drinking water MCL of 50 µg/L MCL. For drinking water standards, the TCLP values are set at 100 times the MCL. If

the drinking water MCL is changed, it is probable that the TCLP limit will also be changed.

There are critics of the TCLP test<sup>2</sup>. They contend that it underestimates leachate from some high alkaline wastes (such as softening residuals) because the buffering capacity of the wastes reduced the leaching potential in the test in a manner that might differ appreciably from that in a landfill. The TCLP test may also not mimic conditions in many non-hazardous industrial landfills. The chelation-facilitated mobility of some wastes is underestimated by the TCLP because of the low chelation activity of the TCLP acetate buffer. Additionally, the TCLP does not account for oxidation/reduction reactions that occur in landfills. This is probably its most serious failure for arsenic containing materials since arsenic is more soluble under reduced conditions.

A different extraction procedure may be needed to test the hazard potential of waste because of the shortcomings of the TCLP. The State of California developed an alternative to the TCLP in the 1980's. This extraction method is called the Waste Extraction Test (WET). The California WET is a more aggressive test than the TCLP because of a different extraction fluid, a longer mixing period, and a higher sample to extraction fluid ratio. Although the California WET method is similar to the TCLP, there are some important differences between the two tests as shown in Table 4-1.

California WET	TCLP
One set extraction fluid. Citric Acid Buffer pH 5.0	Extraction fluid selection depends on sample pH: c. Acetate buffer pH 4.93 d. Acetic Acid solution pH 2.88
Sample to extraction fluid ratio is 1:10	Sample to extraction fluid ratio is 1:20
Does not specify extraction vessel design	Requires extraction bottles made of glass or polypropylene
Requires use of 0.45 µm membrane filter for extract after extraction	Requires use of 0.6 to 0.8 µm glass fiber filter
Uses mechanical shaker for extraction	Requires rotary end over end fashion at 30 rpm
Extraction period of 48 hours	18 hours
Does not require acid digestion after extraction for metals	Requires acid digestion after extraction for metals other than mercury

**Table 4- 1.** Comparison of TCLP and California WET, adapted from Sivia et al, 1991<sup>3</sup>.

The California WET extraction fluid contains citric acid, which is a strong chelating agent. Because of this, the California WET usually extracts higher quantities of most metals than the TCLP test<sup>4</sup>. In previous studies, the California WET was found to extract two to ten times as much arsenic as the TCLP test, even after the TCLP value had

been multiplied by two to account for the different dilution factors in the two tests<sup>3,4</sup>. The California WET extract was also compared to actual municipal solid waste leachate in Hooper's study<sup>4</sup>. The California WET extract generally extracted higher amounts of arsenic than was found in the municipal solid waste leachate. The overestimated level of arsenic extracted by the California WET was attributed to the existence of arsenic bound to iron as arsenopyrite in the waste. The citric acid chelates the iron and therefore solubilizes the arsenic.

A problem with both the TCLP and California WET is the instructions are not specific enough for the testing of residuals. Neither the California WET nor the TCLP specify the conditions for which to analyze wastes such as residuals. Residuals must pass the paint filter test in order to be landfilled, so it is logical that the tests should be performed at the percent solids of passing the paint filter test. However, neither test indicates if the tests should be performed at the percent solids of disposal, at 100% solids, or at some other solids concentration.

Another issue facing the TCLP and California WET is the affect of storage on the disposal of arsenic containing wastes. The TCLP and California WET results may be affected by the age of the waste tested. As wastes age, changes in the redox potential influence changes in the arsenic solubility. Iron solubility is also affected by aging; iron changes from hydroxide forms to oxide forms and becomes less soluble. The changes that occur as waste ages may influence the TCLP and California WET results.

### *Objectives*

Disposal practices are governed by the amount of arsenic present in the waste and the hazard potential of the waste as determined by toxicity testing. Little data is available about the arsenic content or hazard potential of drinking water treatment residuals. In this study the concentration of arsenic in residuals from drinking water treatment plants with high levels of arsenic in their incoming water will be measured.

Little is known about the suitability of the current toxicity testing methods for use on drinking water treatment plant residuals. The application of the TCLP and California WET on drinking water treatment residuals will be investigated. The TCLP and California WET tests will be used to characterize the toxicity of residuals at a dewatered

solid concentration and at a dry solid concentration to establish if the test results are affected by the solid concentrations of the residuals. The TCLP and California WET will also be executed on aging residuals to determine if the test results will be influenced by changes that occur in the soluble arsenic concentration over time.

**METHODS**

Sampling kits and instructions were sent to seven drinking water treatment plants with high arsenic concentrations in their incoming water. Of the seven plants, four used ferric chloride as their coagulant, two used aluminum hydroxide, and one was a softening plant using lime. Residuals from each utility were sent to Virginia Tech for analysis.

The residuals were separated into two jars. The jars were sealed and stored in 20°C controlled temperature in darkness. After two months, one jar of residuals from each utility was tested. After six months, the other jar of residuals from each utility was tested.

The residuals were dewatered to the point of passing the paint filter test using a Whatman #41 filter (20-25 µm pore size). The liquid portions were acid digested using EPA Method 3010a and analyzed using ICP-AES (EPA Method 6010). A hydride generator was used to increase the arsenic detection limit of the instrument.

<b>Residual</b>	<b>Percent Solids After Filtration</b>
Billings	30.4
Great Falls	17.2
Helena	4.1
Indiana	12.6
Lockwood	22.6
Los Angeles	20.0
Louisiana	31.4

**Table 4- 2.**The percent solids of each residual after dewatering by filtration.

The average percent solids of the filtered residual solids was 19.8%, the values for each residual are shown in Table 4-2. Portions of the filtered solid residuals were subjected to both the TCLP and California WET (EPA Method 1311 and California HML SOP 910). The TCLP extract was acid digested using EPA Method 3010a. The TCLP

digestate and the California WET extract were analyzed using ICP-AES (EPA Method 6010) with a hydride generator.

The remaining solid portions of the residuals were oven dried in a 100°C oven. They were then acid digested and analyzed for total iron, aluminum, and arsenic using ICP-AES (EPA Methods 3050b and 6010). The dry solids also underwent TCLP, California WET, and ICP-AES analysis.

## RESULTS AND DISCUSSION

Table 4-3 shows the total arsenic, aluminum, and iron concentrations in each residual, as well as the type of coagulant used and the arsenic to iron or aluminum molar ratio. All of the residuals contain significant amounts of arsenic, considering that current regulations consider “clean” residuals to contain 41 mg/kg arsenic<sup>1</sup>. According to the same regulations, residuals with up to 75 mg/kg arsenic can be land applied in limited amounts. Only the Billings residuals would be land applicable, just meeting the limit.

The Indiana residuals contain an extremely high amount of arsenic. However, the Indiana plant uses ferric chloride as a coagulant and it also contains a great deal of iron. The arsenic to iron molar ratios are in the same range as the other ferric chloride utilities in the study, suggesting that the residual from Indiana contains little other than iron hydroxide.

	As mg/kg	Fe g/kg	Al g/kg	Coagulant	As:Fe molar ratio	As:Al molar ratio
<b>Billings, MT</b>	72.2	74.7	25.5	Ferric Chloride	0.001	-
<b>Great Falls, MT</b>	490.2	14.7	109.2	Aluminum Hydroxide	-	0.002
<b>Helena, MT</b>	421.9	43.7	41.0	Aluminum Hydroxide	-	0.004
<b>Indiana, IN</b>	15729.6	4932.2	1.7	Ferric Chloride	0.002	-
<b>Lockwood, MT</b>	817.7	327.7	11.8	Ferric Chloride	0.002	-
<b>Los Angeles, CA</b>	726.1	139.4	17.0	Ferric Chloride	0.004	-
<b>Louisiana, LS</b>	109.4	2.3	28.3	Lime	-	-

**Table 4-3** The residual arsenic, iron and aluminum concentrations, the coagulant used, and the molar ratios of arsenic to iron or aluminum for each utility.

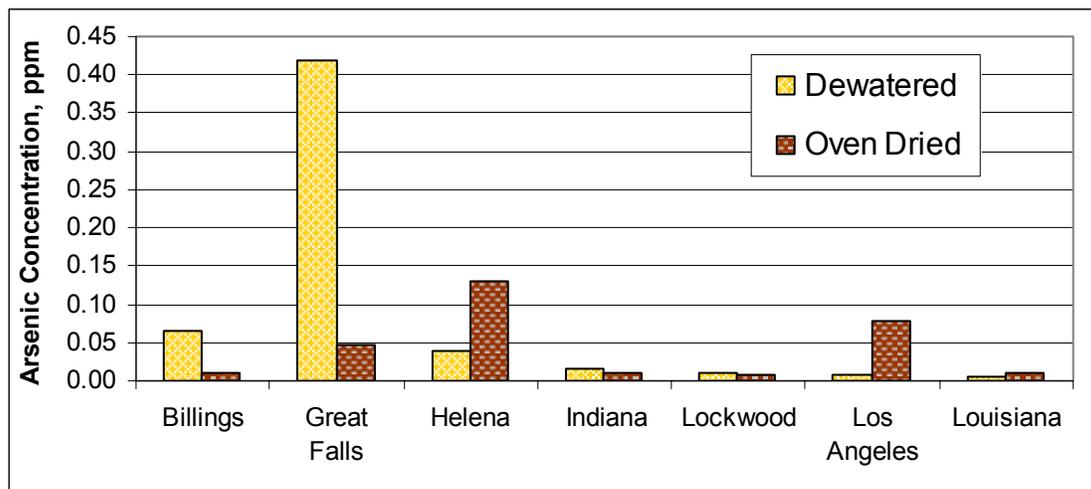
\*Percent solids expressed as a dry weight

### *TCLP and California WET on Dry Residuals and Dewatered Residuals*

After two months of aging, TCLP tests were performed on the residuals; the results are shown in Figure 4-1. The differences in the TCLP values between the residuals at the dewatered solids concentration (approximately 20% solids) and the dry solids concentration (100% solids) are of interest. It could be assumed that the dewatered residuals would produce higher TCLP values because of the presence of soluble arsenic. This was not always the case. Helena, Los Angeles, and Louisiana had higher TCLP values after oven drying than after dewatering.

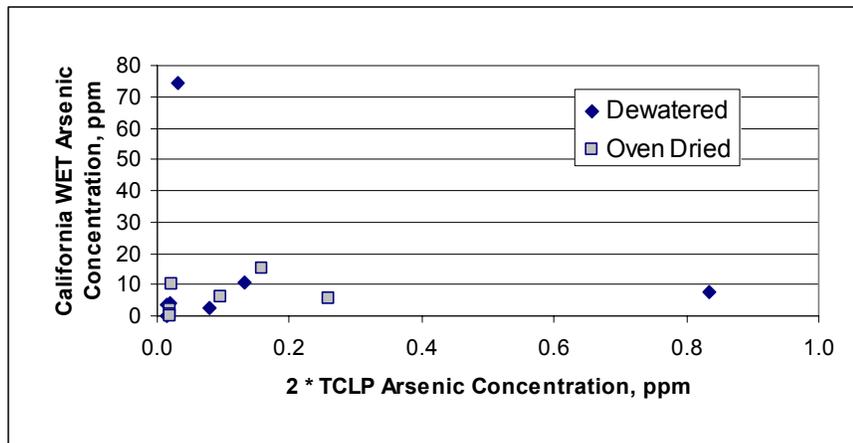
With the current TCLP limit of 5 mg/L, all of the utilities easily pass the test. In fact, the highest TCLP value is 0.42 ppm, which is less than one tenth of the TCLP limit. If the drinking water MCL were changed to 5 ppb arsenic and the TCLP changed to 0.5 ppm, all of these utilities would still pass the TCLP test.

It is interesting to note that the Billings and Lockwood facilities are very close geographically, treating similar water with similar treatments. However, their TCLP results differ. The differences in the TCLP results could be from differences in solids handling after production, or the season of year the residuals were produced. The drinking water treatment plants were instructed to send their freshest residuals, the age of which would vary from plant to plant. Therefore, the residuals collected from Billings may have been produced during a different time of year than the residuals collected from Lockwood.



**Figure 4- 1.** The arsenic concentration in TCLP extracts from drinking water treatment residuals for oven dried residuals (100% solids) and dewatered residuals (approximately 20% solids).

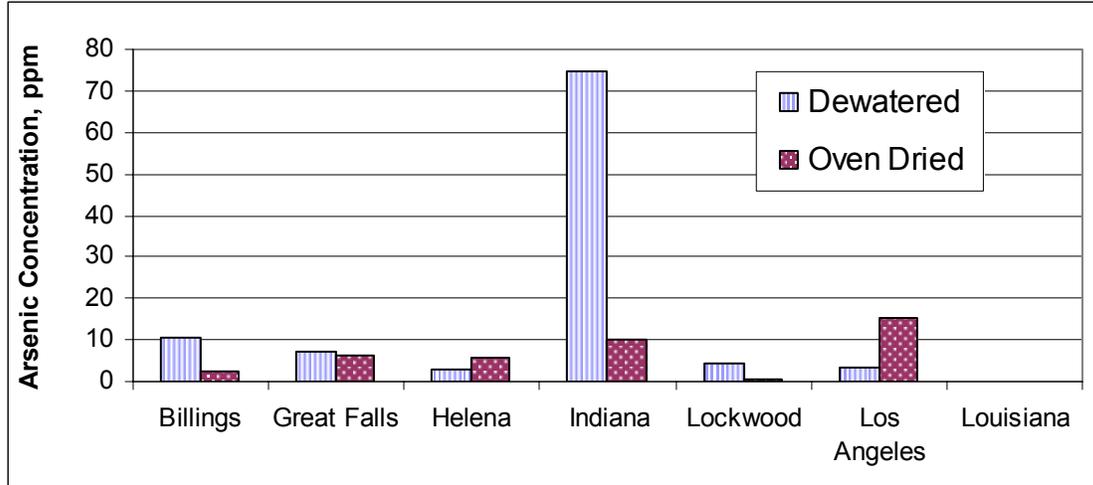
The California WET is a more aggressive test than the TCLP, and the California WET extracted much more arsenic than the TCLP. However, there was no direct relationship between the concentrations of arsenic extracted in the two tests. To directly compare the TCLP and California WET, the TCLP value must be multiplied by two to account for the different dilution factors used in the two tests. After multiplying the TCLP values by two, the California WET extracted an average of approximately 100 times more arsenic. A plot of the California WET versus two times the TCLP value is shown in Figure 4-2.



**Figure 4- 2.** The California WET arsenic concentrations versus two times the TCLP arsenic concentrations.

The California WET results for dewatered solids and dry solids are displayed in Figure 4-3. Similar to the TCLP, the solids concentrations have different affects on the arsenic concentrations extracted from the residuals. With the current California WET limit of 5 mg/L, Billings, Great Falls and Indiana fail the test at dewatered solid concentrations. Great Falls, Helena, Indiana, and Los Angeles fail the test at 100% solids. If changes in the drinking water MCL would result in a California WET regulatory limit of 1 mg/L arsenic, only the Louisiana residuals would pass the test. Because of the strong chelating properties of the California WET citric acid buffer, higher levels of iron are found in the California WET extract than in the TCLP extract. On average, the California WET extract contained over 1300 times the iron as the TCLP extract (after multiplying the TCLP value by two). Arsenic is bound to iron, so more

arsenic consequentially becomes solubilized in the California WET. However, neither the California WET nor the TCLP arsenic and iron concentrations are correlated ( $R^2 < 0.1$ ).



**Figure 4- 3.** Arsenic concentrations of California WET extract of dewatered residuals (at approximately 20% solids) and oven dried residuals (at 100% solids), Louisiana residuals had California WET arsenic concentrations  $< 0.015$  ppm.

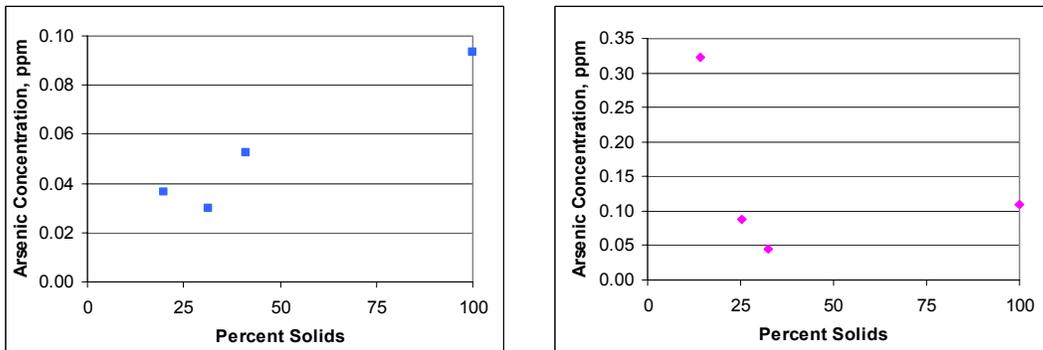
In the dewatered residuals, the California WET value of the Indiana residuals extracted over 2000 times as much arsenic as the TCLP, after multiplying the TCLP value by two. Although the Indiana residuals had an extremely high arsenic concentration in the dry solids, they pass the TCLP test. The Indiana residuals contain high concentrations of arsenic and iron. The strong chelating ability of the California WET extraction fluid must solubilize more of the iron and the iron-bound arsenic than the TCLP extraction fluid.

Helena and Los Angeles residuals exhibited higher California WET and TCLP arsenic concentrations in the dry solids than in the dewatered solids. The Los Angeles residuals might be explained by the age of the residuals. Older residuals may have arsenic and iron present in more crystalline forms, which are more resistant to extraction. As mentioned earlier, the age of the residuals from each utility would vary greatly. The Los Angeles residuals may not be as old as other residuals, and therefore the iron and arsenic would be more susceptible to extraction.

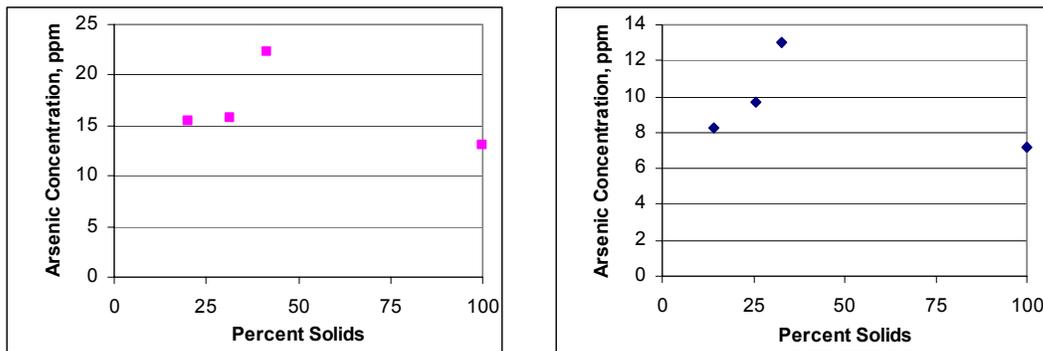
### *TCLP and California WET Values at Varying Percent Solids*

To further examine what mechanisms were behind the changes in the TCLP and California WET test with differing percent solids, the TCLP and California WET were performed on Great Falls and Los Angeles residuals at varying percent solids. The results of these experiments are shown in Figures 4-4 and 4-5.

The Helena residuals, which had increasing California WET results from dewatered to dry percent solids, were very resistant to filtration. They were tested at low percent solids (4.1%). Looking at Figure 4-5, you can imagine that residuals at 4.1% solids would have a low California WET value. This explains why the Helena 100% solids California WET extract was greater than the Helena California WET extract at the lower percent solids.



**Figure 4- 4.** The TCLP test arsenic concentrations at varying percent solids for Los Angeles (Figure 3a) and Great Falls (Figure 3b) residuals.



**Figure 4- 5.** The California WET test results at varying percent solids for the Los Angeles (Figure 4a) and Great Falls (Figure 4b) residuals.

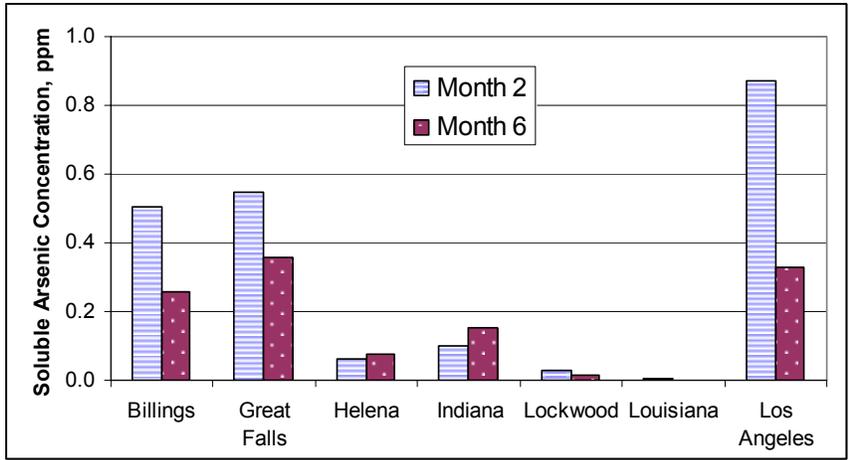
The TCLP results at different percent solids (Figure 4-4) are different for the two residuals tested. In the Los Angeles residual, the TCLP arsenic concentration increases

with increasing percent solids. In Great Falls, the TCLP extract is highest at the lowest percent solids, then decreases, then increases at 100% solids. The soluble arsenic in Los Angeles and Great Falls was measured and found to be very similar. However, the Los Angeles residuals have a solid arsenic concentration 1.5 times higher than the Great Falls solid arsenic concentration (Table 4-2). In the case of the Los Angeles residuals, the TCLP functions to extract the arsenic from the solid portion of the residuals. Increased solids concentrations caused increased amounts of arsenic that may be extracted. The Great Falls TCLP results are cause for speculation.

The California WET arsenic concentrations at varying percent solid concentration (Figure 4-5) are similar for both residuals tested, and can therefore be easily explained. In both the California WET and TCLP, all of the solid concentrations were such that no water would filter out, so the tests were conducted using the same wet weight of residual at each percent solid. As water evaporated off the residuals and the percent solids were increased, the soluble arsenic became more concentrated in the remaining liquid portion. As long as there was still liquid left in the residuals, the California WET arsenic concentrations increased. At 100% solids, no soluble arsenic was present so the arsenic concentration in the California WET decreased as compared to the increasing trend approaching 100% solids. For dry solids, the arsenic in the California WET extract is the arsenic that the citric acid buffer was able to extract from the dry solids.

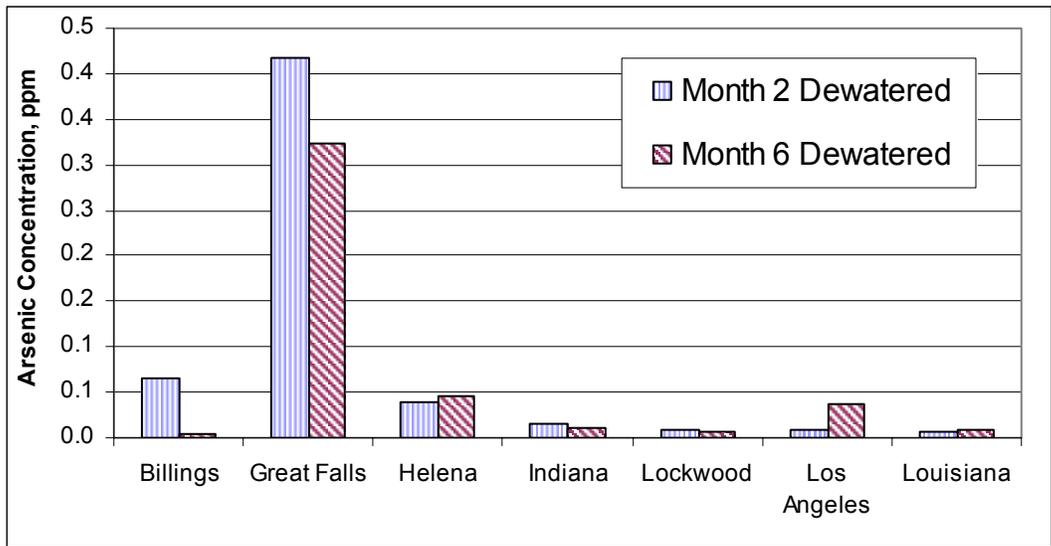
#### *Changes in TCLP and California WET from Aging Residuals*

As residuals age, changes occur in the soluble arsenic concentrations, which may affect TCLP and California WET results. To determine what changes in soluble arsenic were occurring in the residuals, the liquid portions of the residuals were analyzed for arsenic at two months and six months. Figure 4-6 is a graph of the soluble arsenic concentrations in the residuals at two months and six months. The soluble arsenic concentration decreases in Billings, Los Angeles, and Great Falls residuals. Indiana and Helena residuals show an increase in soluble arsenic. Lockwood and Louisiana remain relatively unchanged.

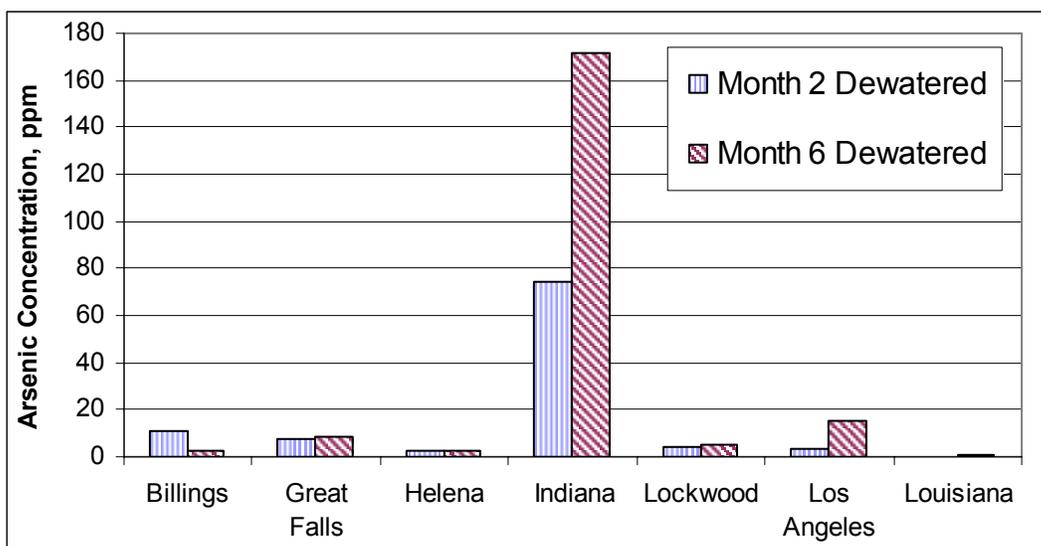


**Figure 4- 6.** The soluble arsenic concentration in residuals after two months and six months of storage.

The TCLP and California WET at dewatered percent solids and dry percent solids were again used to test the residuals after six months of storage. Figures 4-7 and 4-8 show the TCLP and California WET dewatered results after six months of storage as compared to the TCLP and Ca WET dewatered values after 2 months of storage.



**Figure 4- 7.** The residual TCLP arsenic concentrations on dewatered residuals (at approximately 20% solids) after 2 months of storage and after 6 months of storage.



**Figure 4- 8.** The California WET arsenic concentrations after 2 months of residual storage and after 6 months of residual storage on dewatered residuals tested at approximately 20% solids.

The soluble arsenic changes seem to affect the TCLP and California WET results. For example, Billings showed decreases in soluble arsenic content from two month to six months of storage. Billings also showed a decrease in TCLP and California WET dewatered results from 2 months to six months. A decrease in soluble arsenic and a decrease in TCLP and California WET values are also observed in the Great Falls residuals. As the residuals age, the arsenic becomes insoluble and unavailable to the extraction tests. The Los Angeles residuals are the exception to this trend. Although the soluble arsenic concentration decreases from two months to six months, the TCLP and California WET values increase. This might be explained by the age difference of the residuals as previously discussed.

Increased soluble arsenic released in aging waste also affects toxicity testing. The Indiana residuals demonstrated increasing soluble arsenic concentrations. Although the TCLP results remained relatively unchanged, the California WET arsenic concentrations increased dramatically.

## CONCLUSIONS

Residuals from drinking water treatment plants that contain high raw water arsenic concentrations are not suited for land application or monofill disposal. The

residuals contain concentrations of arsenic that exceed the current regulations for these practices.

The disposal option open for arsenic containing residuals is landfilling in municipal solids waste landfills. Only non-hazardous waste may be placed in municipal solid waste landfills, and the current methods for testing the hazard potential of arsenic containing wastes are the TCLP and California WET toxicity tests

Specific methodology for conducting the TCLP and the California WET are not specific and variation in procedure may alter the results of residual testing. Neither of the tests specifies the percent solids at which to test the residuals, even though the percent solids affect the test results. Utilities should examine their residuals and know how the solid concentration affects the toxicity test results. Altering the solid concentration of residuals is an inexpensive and simple way for utilities to lower their TCLP or California WET arsenic concentrations.

Neither the TCLP nor the California WET considers the effects of aging on residuals containing arsenic. As wastes age and the redox potential fluctuates, changes in soluble arsenic occur. These changes are not accounted for in current toxicity testing.

Because disposal of arsenic containing wastes will be mainly in municipal solid waste landfills, alternative toxicity testing methods may be needed. Toxicity testing should have more specific and comprehensible instructions for residuals. The effects of aging should be taken into account to properly identify waste that will release arsenic over time. With the current toxicity testing, there is a very real danger of arsenic contamination from improper disposal of wastes.

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## APPENDIX A. DATA FOR FIGURES

**Figure 3- 1.** Soluble arsenic concentrations (ppm) over six months in the ferric chloride residuals

	Month 0	Month 2	Month 4	Month 6
<b>Billings</b>	0.002	0.503	0.635	0.255
<b>Indiana</b>	0.014	0.102	0.100	0.152
<b>Lockwood</b>	0.000	0.027	0.017	0.014
<b>Los Angeles</b>	0.358	0.870	0.427	0.327

**Figure 3- 2.** Soluble iron concentrations (ppm) over six months in the ferric chloride residuals

	Month 0	Month 2	Month 4	Month 6
<b>Billings</b>	1.57	54.72	55.76	13.60
<b>Indiana</b>	0.50	2.20	2.52	4.93
<b>Lockwood</b>	26.67	25.90	18.92	22.53
<b>Los Angeles</b>	22.01	69.56	57.37	32.23

**Figure 3- 3.** Soluble arsenic vs. soluble iron in the ferric chloride residuals over six months of storage

	Month 0		Month 2		Month 4		Month 6	
	Iron, ppm	Arsenic, ppm						
<b>Billings</b>	1.57	0.002	54.72	0.503	55.76	0.635	13.60	0.252
<b>Indiana</b>	0.50	0.014	2.20	0.102	2.52	0.100	4.93	0.152
<b>Lockwood</b>	26.67	0.000	25.90	0.027	18.92	0.017	22.53	0.014
<b>Los Angeles</b>	22.01	0.358	69.56	0.870	57.37	0.427	32.23	0.327

**Figure 3- 4.** Soluble arsenic concentration versus the change in iron concentration in the ferric chloride residuals over six months of lagoon storage.

	D Iron, ppm	Arsenic, ppm	D Iron, ppm	Arsenic, ppm	D Iron, ppm	Arsenic, ppm
	<b>Billings</b>	53.15	0.503	54.19	0.635	12.03
<b>Indiana</b>	1.69	0.102	2.02	0.100	4.43	0.152
<b>Lockwood</b>	-0.77	0.027	-7.74	0.017	-4.13	0.014
<b>Los Angeles</b>	47.54	0.870	35.36	0.427	10.22	0.327

**Figure 3- 5.** The soluble arsenic concentration (ppm) in aluminium hydroxide residuals over six months.

	Month 0	Month 2	Month 4	Month 6
Great Falls	0.280	0.547	0.354	0.357
Helena	0.031	0.061	0.066	0.078

**Figure 3- 6a.** The soluble aluminum concentrations (ppm) over six months in stored aluminum hydroxide residuals.

	Month 0	Month 2	Month 4	Month 6
Great Falls	0.43	1.44	1.24	2.22
Helena	0.21	3.48	0.59	0.55

**Figure 3-6b.** The soluble iron concentrations for Great Falls and Helena over six months.

	Month 0	Month 2	Month 4	Month 6
Great Falls	0.29	0.21	0.22	0.28
Helena	3.72	18.97	40.69	43.76

**Figure 3- 7.** Soluble arsenic in Los Angeles residuals and Los Angeles residuals with added organic matter.

	Month 0	Month 2	Month 4	Month 6
Los Angeles	0.3584	0.8704	0.4273	0.3266
LA + organic matter	0.4931	1.0451	1.1083	

**Figure 3- 8.** The soluble arsenic concentrations in the lagoon residuals vs. the redox potentials.

	Month 0		Month 2		Month 4		Month 6	
	Redox Potential, mV	Arsenic, ppm						
Billings	53	0.002	-167	0.503	-176	0.635	-195	0.255
Great Falls	52	0.280	-112	0.547	-133	0.354	-145	0.357
Helena	116	0.031	-61	0.061	-95	0.066	-149	0.078
Indiana	287	0.014	194	0.102	214	0.100	187	0.152
Lockwood	-81	0.000	-110	0.027	-136	0.017	-157	0.014
Louisiana	33	0.000	178	0.005	107	0.003	132	0.000
Los Angeles	-116	0.358	-150	0.870	-142	0.427	-171	0.327

**Figure 3- 9.** Soluble iron in residual lagoons vs. redox potentials.

	Month 0		Month 2		Month 4		Month 6	
	Redox Potential, mV	Iron, ppm						
Billings	53	1.57	-167	54.72	-176	55.76	-195	13.60
Great Falls	52	0.29	-112	0.21	-133	0.22	-145	0.28
Helena	116	3.72	-61	18.97	-95	40.69	-149	43.76
Indiana	287	0.50	194	2.20	214	2.52	187	4.93
Lockwood	-81	26.67	-110	25.90	-136	18.92	-157	22.53
Louisiana	33	0.46	178	0.03	107	0.04	132	0.00
Los Angeles	-116	22.01	-150	69.56	-142	57.37	-171	32.23

**Figure 3- 10.** TCLP arsenic concentrations of the residuals after 2 months of storage and 6 months of storage.

	Billings	Great Falls	Helena	Indiana	Lockwood	Los Angeles	Louisiana
Month 2 at 20% Solids	0.065	0.418	0.040	0.015	0.010	0.008	0.007
Month 6 at 20% Solids	0.005	0.323	0.045	0.012	0.006	0.037	0.009

**Figure 3- 11.** California WET arsenic concentrations of the residuals after two months of storage and after 6 months of storage.

	Billings	Great Falls	Helena	Indiana	Lockwood	Los Angeles	Louisiana
Month 2 at 20% Solids	10.720	7.370	2.730	74.670	4.120	3.570	0.150
Month 6 at 20% Solids	2.579	8.213	2.476	171.823	4.824	15.502	0.437

**Figure 3- 12.** The arsenic concentrations (ppm) of sandbed leachates over a three week period.

	Control	pH 5.8 Light	pH 5.8 Dark	pH 4.8	pH 3.8
Week 1	0.008	0.016	0.016	0.009	0.010
Week 2	0.003	0.007	0.004	0.004	0.006
Week 3	0.003	0.006	0.004	0.004	0.006

**Figure 4- 1.** The arsenic concentration (ppm) in TCLP extracts from drinking water treatment residuals at dewatered and dry percent solids.

	Billings	Great Falls	Helena	Indiana	Lockwood	Los Angeles	Louisiana
20% Solids	0.065	0.418	0.040	0.015	0.010	0.008	0.007
100 % Solids	0.010	0.048	0.130	0.010	0.009	0.079	0.010

**Figure 4- 2.** The California WET arsenic concentrations (ppm) versus two times the TCLP arsenic concentrations (ppm).

	2 * TCLP Dewatered	Ca WET Dewatered	2 * TCLP Oven Dried	Ca WET Oven Dried
Billings	0.132	10.72	0.018	2.26
Great Falls	0.835	7.37	0.096	6.11
Helena	0.079	2.73	0.260	5.61
Indiana	0.032	74.67	0.022	10.23
Lockwood	0.019	4.12	0.018	0.26
Los Angeles	0.016	3.57	0.158	15.24
Louisiana	0.013	0.15	0.019	0.05

**Figure 4- 3.** Arsenic concentrations (ppm) of California WET extract on dewatered residuals and dry residuals.

	Billings	Great Falls	Helena	Indiana	Lockwood	Los Angeles	Louisiana
Dewatered	10.72	7.37	2.73	74.67	4.12	3.57	0.15
Oven Dried	2.26	6.11	5.61	10.23	0.26	15.24	0.05

**Figure 4- 4.** The TCLP test arsenic concentrations (ppm) at varying percent solids for Los Angeles (Figure 4-4a) and Great Falls (Figure 4-4b) residuals.

	Percent Solids	Arsenic, ppm
Los Angeles	100	0.093
	41.32	0.053
	31.56	0.030
	19.9	0.037

	Percent Solids	Arsenic, ppm
Great Falls	100	0.109
	32.63	0.044
	25.46	0.088
	14.16	0.322

**Figure 4- 5.** The California WET test results (arsenic ppm) at varying percent solids for the Los Angeles (Figure 4-5a) and Great Falls (Figure 4-5b) residuals.

	Percent Solids	Arsenic, ppm
Los Angeles	100	13.105
	41.32	22.309
	31.56	15.691
	19.9	15.502

	Percent Solids	Arsenic, ppm
Great Falls	100	7.183
	32.63	13.012
	25.46	9.710
	14.16	8.213

**Figure 4- 6.** The soluble arsenic concentrations (ppm) in residuals after two months and six months of storage.

	Month 2	Month 6
Billings	0.503	0.255
Great Falls	0.547	0.357
Helena	0.061	0.078
Indiana	0.102	0.152
Lockwood	0.027	0.014
Louisiana	0.005	0.000
Los Angeles	0.870	0.327

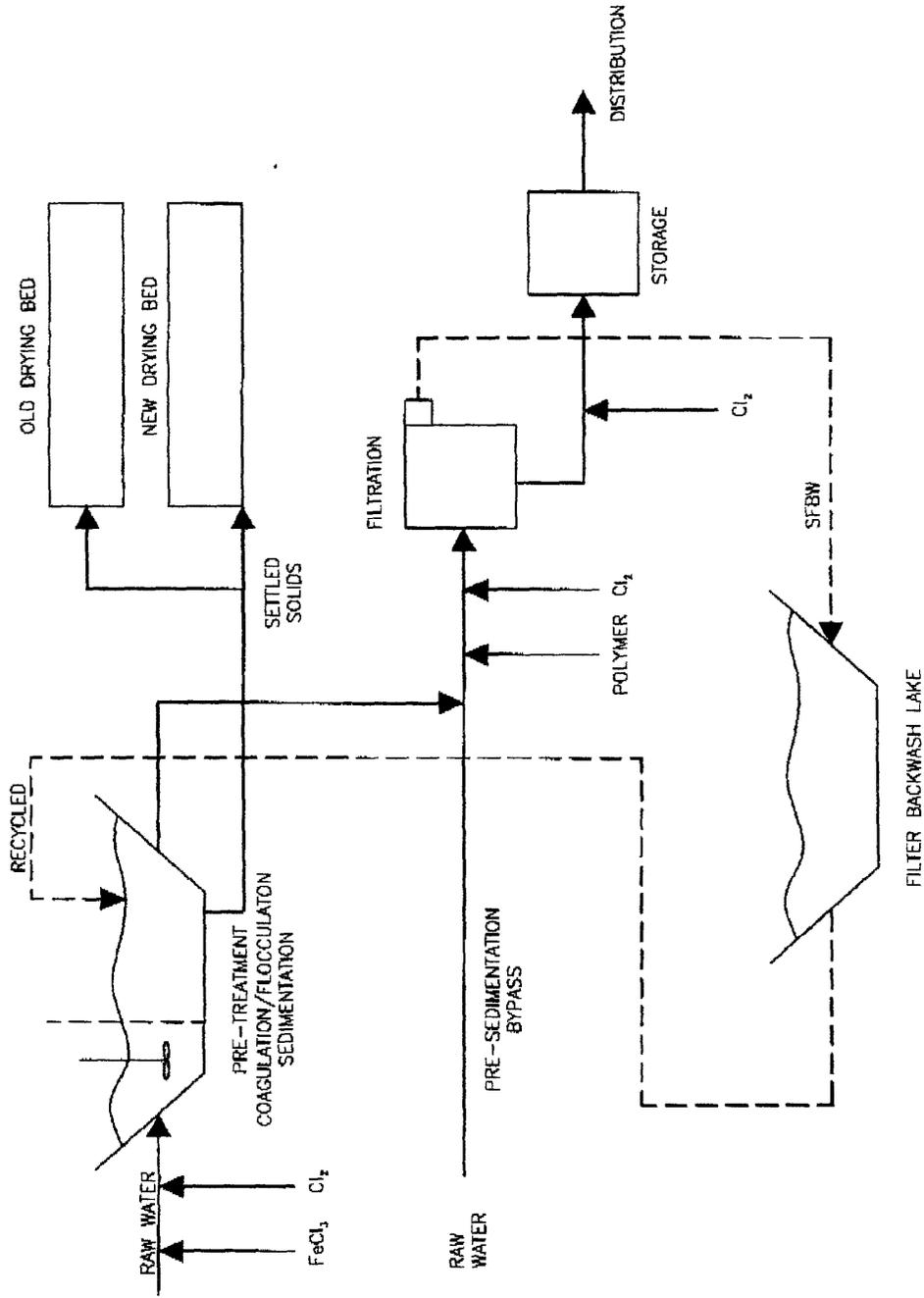
**Figure 4- 7.** The residual TCLP arsenic concentrations (ppm) on dewatered residuals after 2 months of storage and after 6 months of storage.

	Billings	Great Falls	Helena	Indiana	Lockwood	Los Angeles	Louisiana
Month 2 at 20% Solids	0.065	0.4175	0.04	0.015	0.0095	0.008	0.0065
Month 6 at 20% Solids	0.0051	0.3225	0.0454	0.0118	0.0056	0.0367	0.009

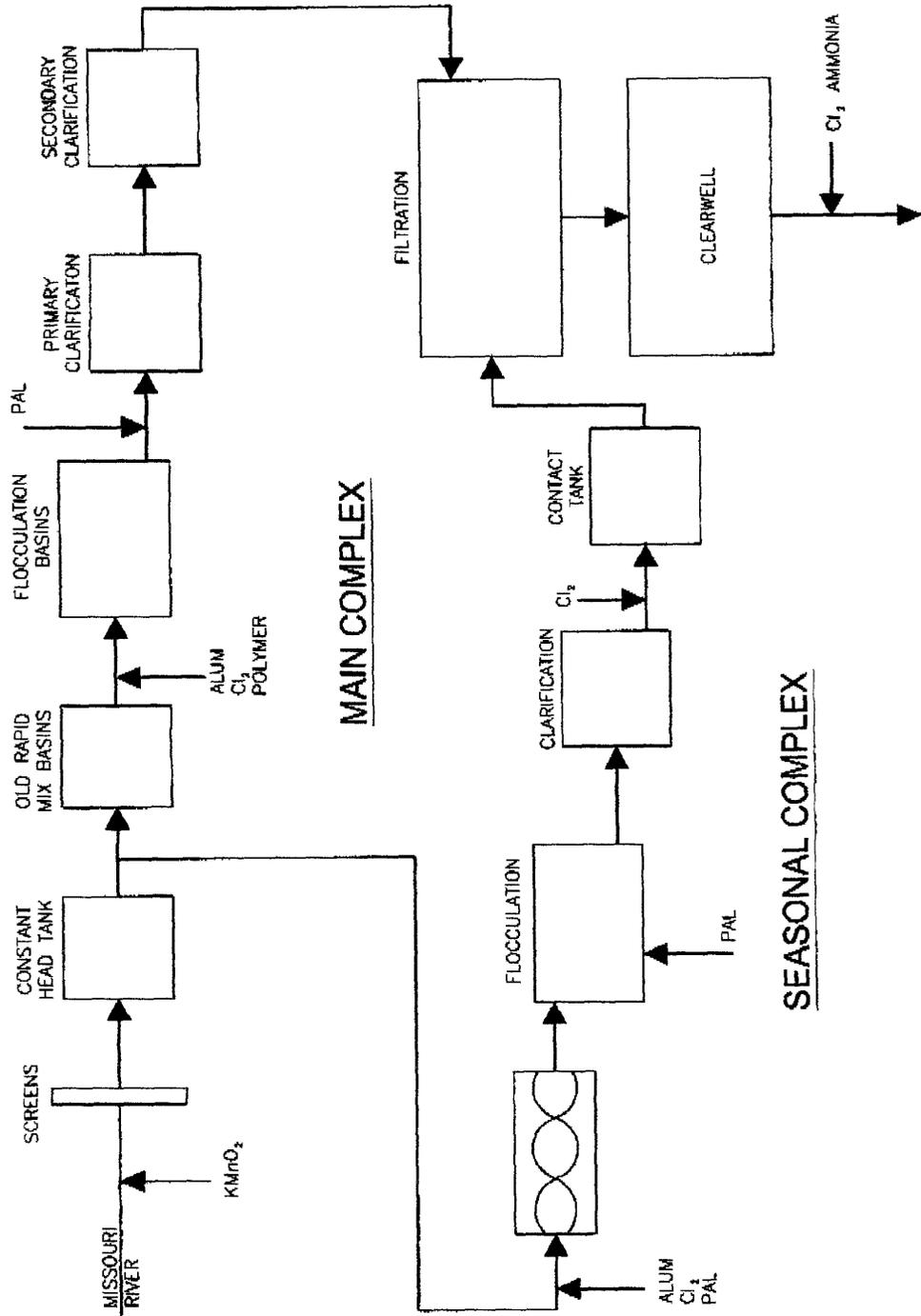
**Figure 4- 8.** The California WET arsenic concentrations (ppm) after 2 months of residual storage and after 6 months of residual storage on dewatered residuals.

	Billings	Great Falls	Helena	Indiana	Lockwood	Los Angeles	Louisiana
Month 2 at 20% Solids	10.72	7.37	2.73	74.67	4.12	3.57	0.15
Month 6 at 20% Solids	2.58	8.21	2.48	171.82	4.82	15.50	0.44

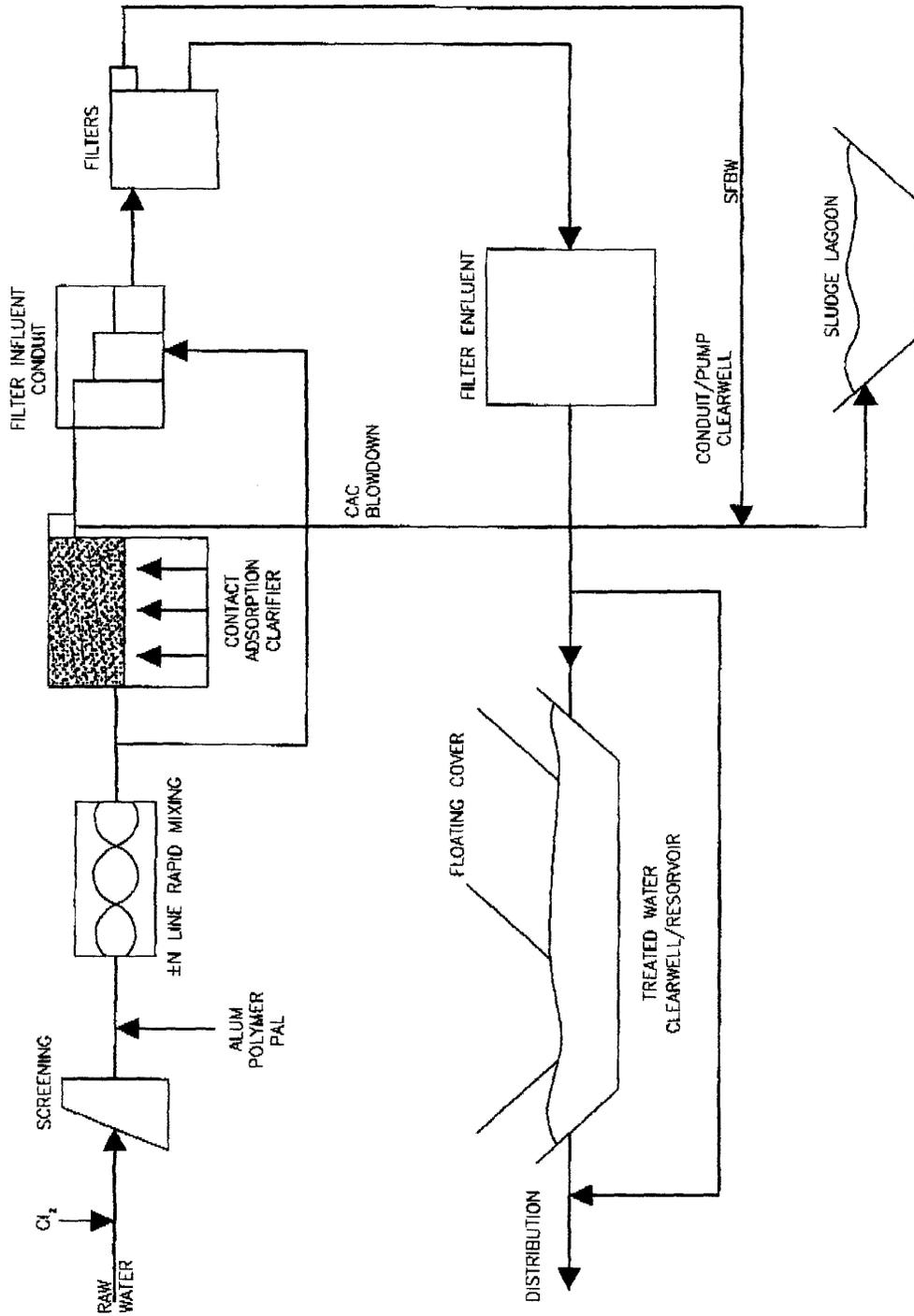
## APPENDIX B. TREATMENT PLANT SCHEMATICS



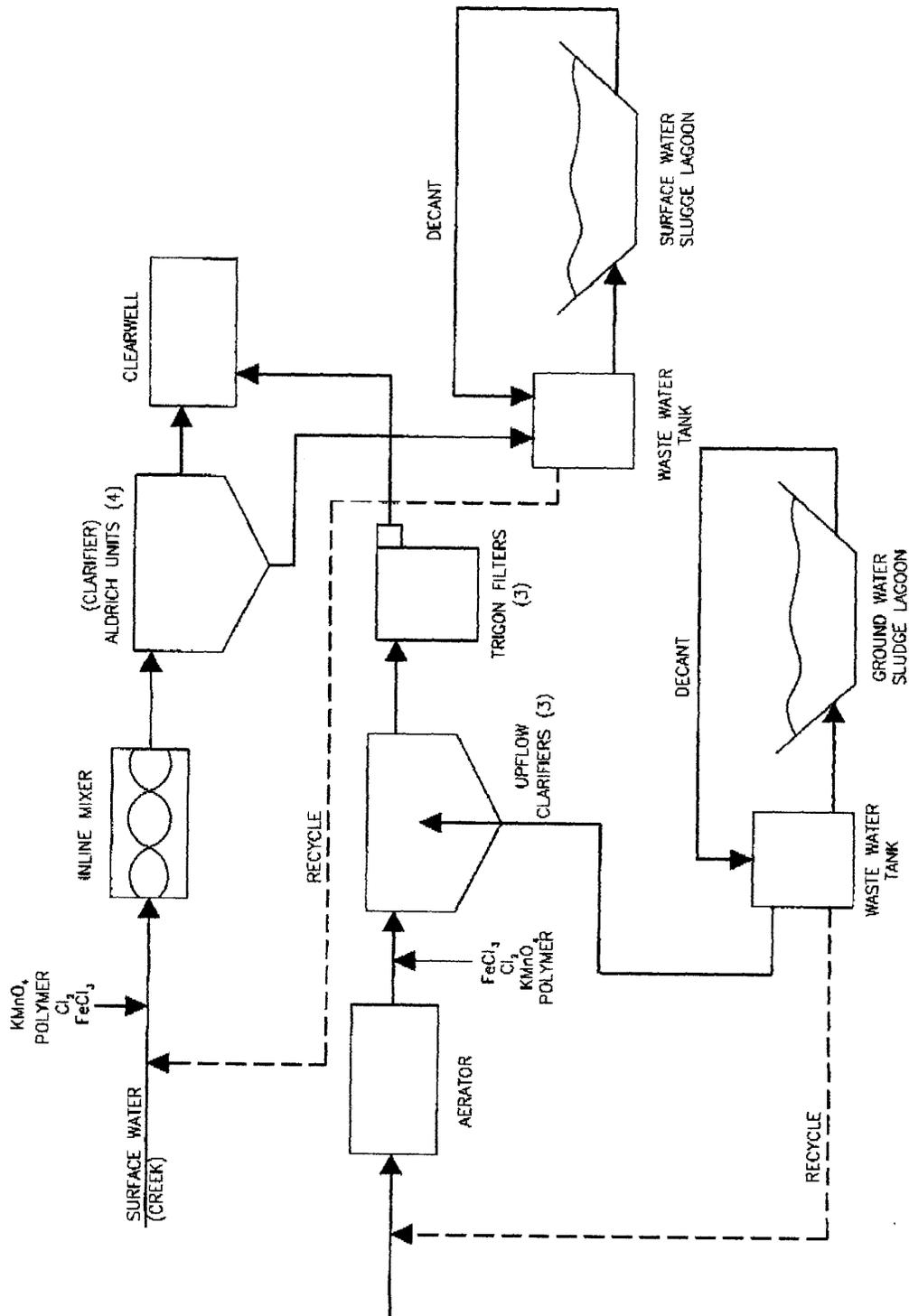
City of Billings WTR Process Schematic  
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City of Great Falls, Process Schematic  
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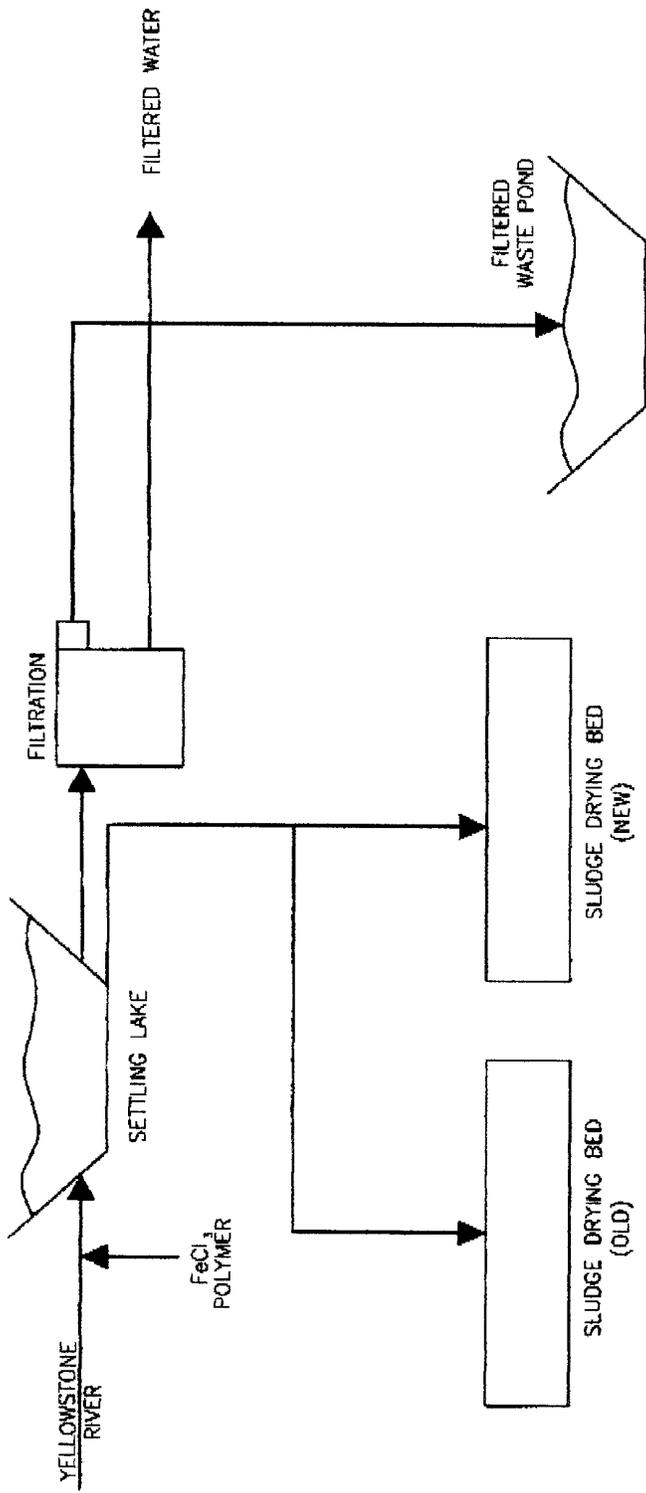


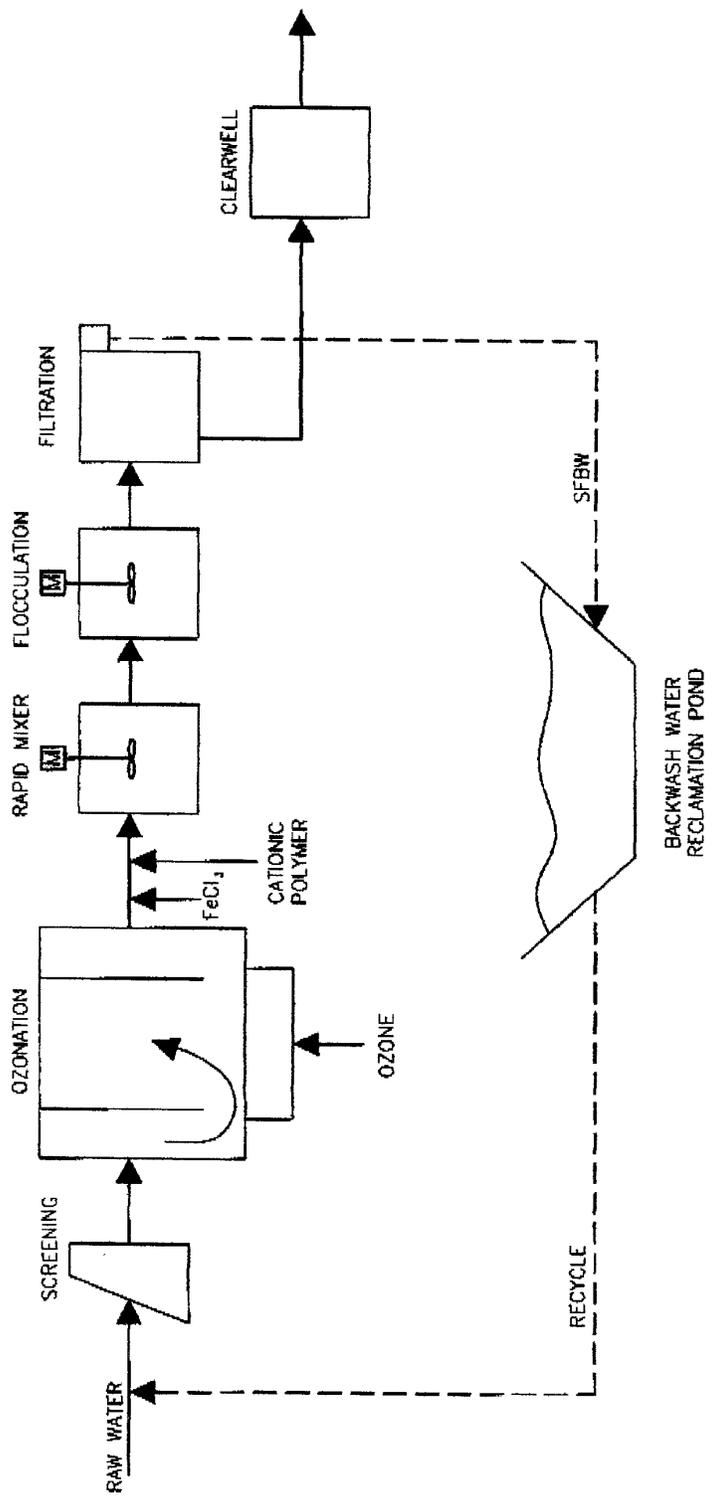
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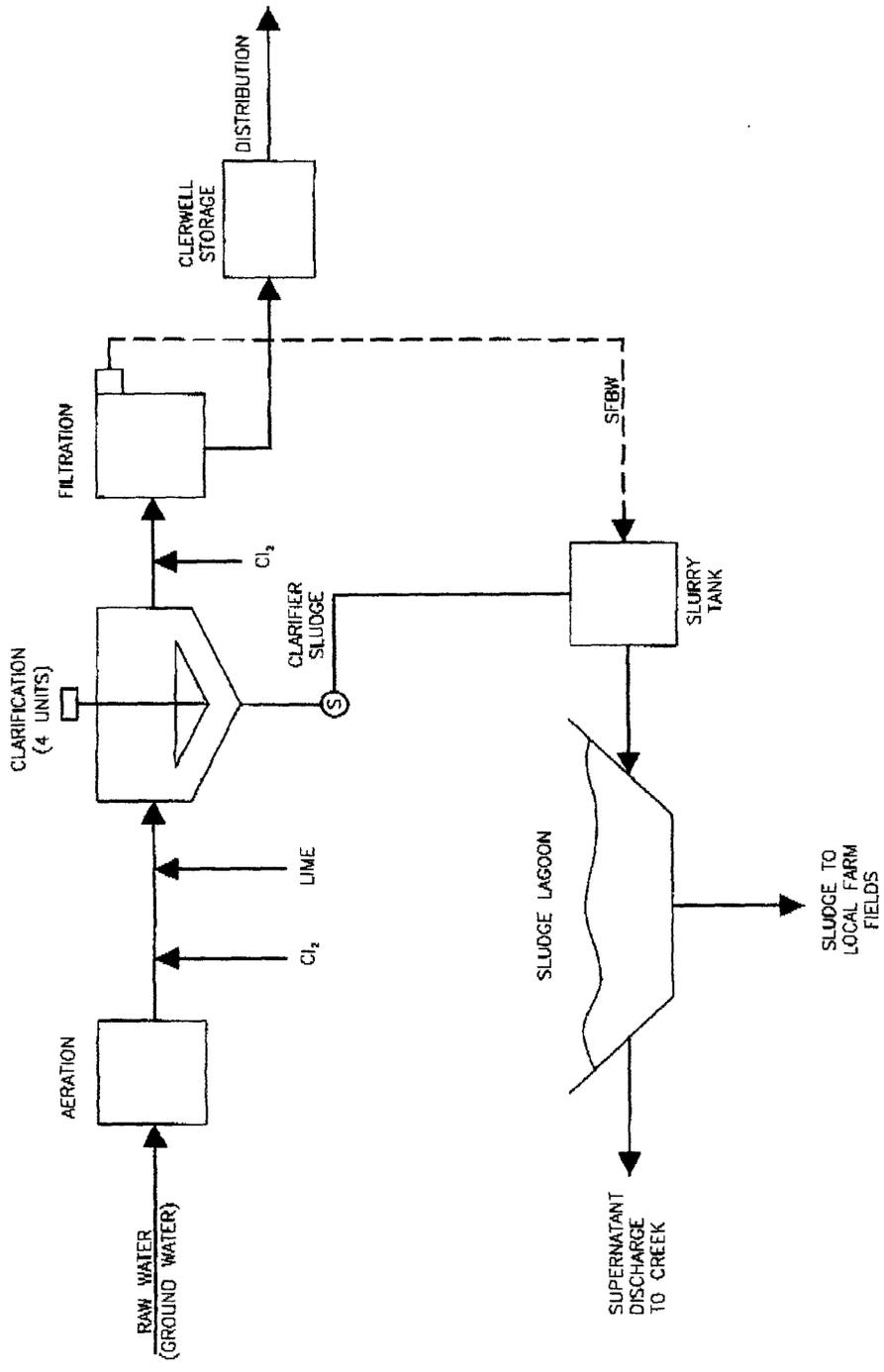
Indiana American, Kokomo, In. WTP Process Schematic

orsrenpr1.dwg





Los Angeles Aqueduct WTP, Process Schematic  
 orsrempri.dwg



New Iberia Process Schematic  
 erarempr1.dwg

## **VITA**

Cortney H. Itle was born in Loretto, Pennsylvania on September 9, 1976. She attended Penn Cambria High School and graduated in 1995. Upon graduation she enrolled in the Pennsylvania State University. She earned her Bachelor's degree in 1999 in Environmental Resource Management with a minor in Environmental Engineering. In August 1999 she began work on her Master's degree at Virginia Polytechnic Institute and State University.