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CORRECTIVE MEASURES STUDY REPORT AREA OF CONCERN 617 (AOC 617) ZONE F  
WITH TRANSMITTAL CNC CHARLESTON SC  
2/27/2002  
CH2M HILL

# CORRECTIVE MEASURES STUDY REPORT

## Area of Concern 617, Zone F



***Charleston Naval Complex  
North Charleston, South Carolina***



SUBMITTED TO  
***U.S. Navy Southern Division  
Naval Facilities Engineering Command***

PREPARED BY  
***CH2M-Jones***

*February 2002*

Revision 0  
Contract N62467-99-C-0960  
158814.ZF.PR.03



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February 27, 2002

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Division of Hazardous and Infectious Wastes  
South Carolina Department of Health and  
Environmental Control  
Bureau of Land and Waste Management  
2600 Bull Street  
Columbia, SC 29201

Re: Corrective Measures Study Report (Revision 0) – AOC 617, Zone F

Dear Mr. Scaturo:

Enclosed please find two copies of the Corrective Measures Study Report (Revision 0) for AOC 617 in Zone F of the Charleston Naval Complex (CNC). This report has been prepared pursuant to agreements by the CNC BRAC Cleanup Team for completing the RCRA Corrective Action process.

Please do not hesitate to contact me at 352/335-5877, extension 2280, if you have any questions or comments.

Sincerely,

CH2M HILL

Dean Williamson, P.E.

cc: Tim Frederick/Gannett Fleming, Inc. w/att  
Rob Harrell/Navy, w/att  
Gary Foster/CH2M HILL, w/att

# CORRECTIVE MEASURES STUDY REPORT

## Area of Concern 617, Zone F



***Charleston Naval Complex  
North Charleston, South Carolina***



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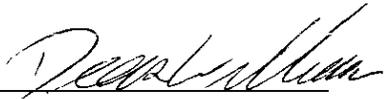
Revision 0  
Contract N62467-99-C-0960  
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# Certification Page for Corrective Measures Study Report (Revision 0) — AOC 617, Zone F

I, Dean Williamson, certify that this report has been prepared under my direct supervision. The data and information are, to the best of my knowledge, accurate and correct, and the report has been prepared in accordance with current standards of practice for engineering.

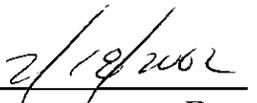
South Carolina

P.E. No. 21428



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Dean Williamson, P.E.



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Date

# 1 Contents

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2 Section	Page
3 <b>Acronyms and Abbreviations</b> .....	vii
4 <b>1.0 Introduction</b> .....	1-1
5     1.1 Corrective Measures Study Report Purpose and Scope .....	1-1
6     1.2 Background Information.....	1-2
7         1.2.1 Facility Description .....	1-2
8         1.2.2 Site History.....	1-2
9         1.2.3 COC Summary .....	1-2
10        1.2.4 Summary of Conclusions and Recommendations from the RFI Report Addendum/CMS Work Plan (CH2M-Jones, 2001).....	1-4
12     1.3 Overall Approach for Selecting Candidate Corrective Measure Alternatives for AOC 617 .....	1-4
14     1.4 Report Organization .....	1-5
15 Figure 1-1 Location of AOC 617, Zone F within the CNC.....	1-6
16 Figure 1-2 Aerial Photograph of AOC 617 .....	1-7
17 Figure 1-3 RFI Addendum Soil Sample Locations .....	1-8
18 Figure 1-4 RFI Addendum Groundwater Well Locations.....	1-9
19 Figure 1-5 Approximate Area of Zinc Plume in Groundwater .....	1-10
20 <b>2.0 Remedial Goal Objectives and Evaluation Criteria</b> .....	2-1
21     2.1 Remedial Action Objectives.....	2-1
22     2.2 Media Cleanup Standards .....	2-1
23     2.3 Evaluation Criteria .....	2-2
24 <b>3.0 Description of Candidate Corrective Measure Alternatives</b> .....	3-1
25     3.1 Introduction .....	3-1
26     3.2 Alternative 1: Natural Attenuation with Land Use Controls .....	3-1
27         3.2.1 Description of Alternative .....	3-1
28         3.2.2 Key Uncertainties.....	3-2
29         3.2.3 Other Considerations.....	3-2
30     3.3 Alternative 2: In-situ Stabilization/Precipitation.....	3-3
31         3.3.1 Description of Alternative .....	3-3

# 1 Contents, Continued

---

2	3.3.2 Key Uncertainties.....	3-3
3	3.3.3 Other Considerations.....	3-3
4	3.4 Alternative 3: Groundwater Extraction, Treatment, and Discharge to the	
5	Sanitary Sewer .....	3-4
6	3.4.1 Description of Alternative.....	3-4
7	3.4.2 Key Uncertainties.....	3-4
8	3.4.3 Other Considerations.....	3-6
9	<b>4.0 Evaluation and Comparison of Corrective Measure Alternatives .....</b>	<b>4-1</b>
10	4.1 Alternative 1: Natural Attenuation with Land Use Controls .....	4-1
11	4.1.1 Protection of Human Health and the Environment.....	4-1
12	4.1.2 Attain MCS.....	4-1
13	4.1.3 Control the Source of Releases .....	4-1
14	4.1.4 Compliance with Applicable Standards for the Management of	
15	Generated Wastes.....	4-2
16	4.1.5 Other Factors (a) Long-term Reliability and Effectiveness .....	4-2
17	4.1.6 Other Factors (b) Reduction in the Toxicity, Mobility, or Volume of	
18	Wastes .....	4-2
19	4.1.7 Other Factors (c) Short-term Effectiveness .....	4-2
20	4.1.8 Other Factors (d) Implementability .....	4-2
21	4.1.9 Other Factors (e) Cost.....	4-2
22	4.2 Alternative 2: In-situ Stabilization/Precipitation.....	4-2
23	4.2.1 Protection of Human Health and the Environment.....	4-3
24	4.2.2 Attain MCS.....	4-3
25	4.2.3 Control the Source of Releases .....	4-3
26	4.2.4 Compliance with Applicable Standards for the Management of	
27	Generated Wastes.....	4-3
28	4.2.5 Other Factors (a) Long-term Reliability and Effectiveness .....	4-3
29	4.2.6 Other Factors (b) Reduction in the Toxicity, Mobility, or Volume of	
30	Wastes .....	4-3
31	4.2.7 Other Factors (c) Short-term Effectiveness .....	4-4
32	4.2.8 Other Factors (d) Implementability .....	4-4
33	4.2.9 Other Factors (e) Cost.....	4-4

# 1 Contents, Continued

---

2	4.3	Alternative 3: Groundwater Extraction, Treatment, and Discharge to the	
3		Sanitary Sewer .....	4-4
4	4.3.1	Protection of Human Health and the Environment .....	4-5
5	4.3.2	Attain MCS.....	4-5
6	4.3.3	Control the Source of Releases .....	4-5
7	4.3.4	Compliance with Applicable Standards for the Management of	
8		Generated Wastes.....	4-5
9	4.3.5	Other Factors (a) Long-term Reliability and Effectiveness .....	4-5
10	4.3.6	Other Factors (b) Reduction in the Toxicity, Mobility, or Volume of	
11		Wastes .....	4-5
12	4.3.7	Other Factors (c) Short-term Effectiveness .....	4-6
13	4.3.8	Other Factors (d) Implementability .....	4-6
14	4.3.9	Other Factors (e) Cost .....	4-6
15	4.4	Comparative Ranking of Corrective Measure Alternatives.....	4-6
16		Table 4-1 Ranking of Corrective Measure Alternatives .....	4-7
17	5.0	Recommended Corrective Measure Alternative .....	5-1
18	6.0	References.....	6-1
19			
20		<b>Appendices</b>	
21	A	Groundwater Elevation Contours ( <i>Zone F RFI Report, Revision 0</i> )	
22	B	Technology Evaluation Report – Remediation of Metals-contaminated Soils and	
23		Groundwater (GWR TAC, 1997)	
24	C	Natural Attenuation Modeling (Freeze and Cherry, 1979)	
25	D	Cost Estimates for Corrective Measure Alternatives	

# 1 Acronyms and Abbreviations

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2	AOC	area of concern
3	BRAC	Base Realignment and Closure Act
4	CA	corrective action
5	CMS	corrective measures study
6	CNC	Charleston Naval Complex
7	COC	chemical of concern
8	COPC	chemical of potential concern
9	CPW	Charleston Public Works
10	DAF	dilution attenuation factor
11	EnSafe	EnSafe, Inc.
12	EPA	U.S. Environmental Protection Agency
13	ft bls	feet below land surface
14	gpm	gallon per minute
15	GWRTAC	Ground-Water Remediation Technologies Analysis Center
16	HI	hazard index
17	ILCR	Incremental Lifetime Cancer Risk
18	µg/L	microgram per liter
19	LUC	land use control
20	LUCMP	land use control management plan
21	MCL	maximum contaminant level
22	MCS	media cleanup standard
23	NAVBASE	Naval Base
24	NFA	no further action
25	ORB	oxidation-reduction potential
26	PCB	polychlorinated biphenyl
27	PRB	permeable reactive barrier
28	PRG	preliminary remediation goal
29	PVC	polyvinyl chloride
30	RAO	remedial action objective

# 1 **Acronyms and Abbreviations, Continued**

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2	RBC	risk-based concentration
3	RCRA	Resource Conservation and Recovery Act
4	RFI	RCRA Facility Investigation
5	RGO	remedial goal option
6	SCDHEC	South Carolina Department of Health and Environmental Control
7	SSL	soil screening level
8	SVOC	semivolatile organic compound
9	VOC	volatile organic compound
10	UST	underground storage tank



# 1.0 Introduction

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In 1993, Naval Base (NAVBASE) Charleston was added to the list of bases scheduled for closure as part of the Defense Base Realignment and Closure Act (BRAC), which regulates closure and transition of property to the community. The Charleston Naval Complex (CNC) was formed as a result of the dis-establishment of the Charleston Naval Shipyard and NAVBASE on April 1, 1996.

Corrective Action (CA) activities are being conducted under the Resource Conservation and Recovery Act (RCRA), with the South Carolina Department of Health and Environmental Control (SCDHEC) as the lead agency for CA activities at the CNC. All RCRA CA activities are performed in accordance with the Final Permit (Permit No. SC0 170 022 560). In April 2000, CH2M-Jones was awarded a contract to provide environmental investigation and remediation services at the CNC.

A RCRA Facility Investigation (RFI) Report Addendum and Corrective Measures Study (CMS) Work Plan were prepared for Area of Concern (AOC) 617 in Zone F of the CNC (CH2M-Jones, 2001). The RFI Report Addendum and CMS Work Plan presented the remedial action objectives (RAOs) and media cleanup standards (MCSs) proposed for AOC 617, and was approved by the U.S. Environmental Protection Agency (EPA) Region IV on behalf of SCDHEC in December 2001. This same report recommended no further action (NFA) for AOC 616, which is located approximately 50 feet north of AOC 617; the NFA recommendation was also approved by EPA Region IV and SCDHEC. This CMS report has been prepared by CH2M-Jones to complete the next stage of the CA process for AOC 617.

## 1.1 Corrective Measures Study Report Purpose and Scope

This CMS report evaluates corrective measure alternatives for zinc-contaminated groundwater at AOC 617. Zinc in groundwater was the only chemical of concern (COC) identified for AOC 617 in the RFI Report Addendum. Figure 1-1 illustrates the location of AOC 617 within Zone F. The insert on Figure 1-1 shows the location of Zone F within the CNC. Figure 1-2 is an aerial photograph showing the layout of AOC 617.

This CMS report consists of: 1) the identification of a set of corrective measure alternatives that are considered to be technically appropriate for addressing zinc-contaminated groundwater; 2) an evaluation of the alternatives using standard criteria from EPA RCRA

1 guidance; and 3) the selection of a recommended (preferred) corrective measure alternative  
2 for the site.

## 3 **1.2 Background Information**

4 This section of the CMS report presents background information on the facility, site history,  
5 and a summary of the nature and extent of the COCs at the site. This information is essential  
6 to the understanding of the remedial goal options (RGOs), MCSs, and ultimately the  
7 evaluation of corrective measure alternatives for AOC 617. Additional information on the  
8 site and hydrogeology in the Zone F area of the CNC is provided in the *Zone F RFI Report,*  
9 *Revision 0* (EnSafe Inc. [EnSafe], 1999).

### 10 **1.2.1 Facility Description**

11 As shown in Figures 1-1 and 1-2 of this report, AOC 617 is currently paved. AOC 617 is  
12 located in an industrial area east of Hobson Avenue. The CNC Reuse Plan identifies this  
13 area for industrial land use. The City of North Charleston zoning for this site is M-2, for  
14 marine industrial use.

### 15 **1.2.2 Site History**

16 AOC 617 is the site of a former galvanizing plant, designated Building 1176, which operated  
17 from the early 1940s to approximately 1985. Shortly thereafter, Building 1176 was  
18 demolished to facilitate the expansion of Building 69, which is a shipping and supply  
19 warehouse located immediately south of AOC 617. As stated earlier, the site is currently  
20 paved and is used as an access area for shipping operations. Historical drawings also  
21 indicate that this area was paved during Building 1176 operation.

22 Information regarding specific details of historical galvanizing operations conducted at the  
23 site is limited. Available records indicate the former presence of a single 3,000-gallon  
24 underground storage tank (UST) used for chemical storage. Historical records also indicate  
25 the presence of a series of large (approximately 15 by 20 ft) rectangular tanks within the  
26 building, which were used for acid, caustic, chemical storage, and process use. These tanks  
27 were apparently removed in conjunction with the demolition of the building. There is no  
28 record of a release(s) from any of the aforementioned tanks.

### 29 **1.2.3 COC Summary**

30 Over three sampling events during the RFI, EnSafe and CH2M-Jones sampled surface (0 to 1  
31 ft below land surface [ft bls]) and subsurface (3 to 5 ft bls) soil at the seven locations shown

1 in Figure 1-3. Soil samples were analyzed for volatile organic compounds (VOCs),  
2 semivolatile organic compounds (SVOCs), metals, pesticides/polychlorinated biphenyls  
3 (PCBs), and cyanide. Detailed information on the analytical results and the screening of  
4 those results for the determination of COCs can be found in the *Zone F RFI Report, Revision 0*  
5 (*EnSafe, 1997*), and the *RFI Report Addendum and CMS Work Plan for AOC 616/617, Zone F,*  
6 *Revision 0* (CH2M Jones 2001). No surface or subsurface soil COCs were identified for AOC  
7 617.

8 Although the subsurface soil zinc concentration at F617SB003 was greater than the  
9 background range of concentrations for Zones F and G, the zinc concentration was less than  
10 the EPA soil screening level (SSL) (at a dilution attenuation factor [DAF]=10), which  
11 indicates that the subsurface soil in this area is not a source for the zinc in groundwater, and  
12 does not require remedial action.

13 Four groundwater wells were installed at AOC 617 over a period of five years. The locations  
14 of these wells are shown in Figure 1-4. Groundwater samples were analyzed for metals,  
15 PCBs, and SVOCs. Results of groundwater analyses were compared to the screening criteria,  
16 and the chemicals of potential concern (COPCs) that were identified included aluminum,  
17 arsenic, cadmium, cobalt, manganese, nickel, thallium, and zinc. The concentrations of these  
18 metals were reviewed and compared to appropriate screening criteria in the RFI Report  
19 Addendum. Based on this analysis of the COPC concentrations, the only groundwater COC  
20 identified at AOC 617 was zinc. Zinc exceeded the applicable criteria during more recent  
21 sampling only in monitoring well F617GW003.

22 Potentiometric contours of groundwater underneath AOC 617 are shown in Figure 10.9-2  
23 and 10.9-3 of the *Zone F RFI Report, Revision 0*, and are included in this report in Appendix  
24 A. They illustrate the shallow groundwater at low and high tides, respectively. Under either  
25 of these conditions, monitoring well F617GW002 is located upgradient of the site, and  
26 monitoring well F617GW001 is generally side/crossgradient of the site. The first sampling  
27 event conducted at F617GW002 showed several elevated concentrations of metals; the  
28 subsequent four sampling events performed at this well presented concentrations that were  
29 substantially lower (see Appendix A). The monitoring well that appears to be within the  
30 potential source area for zinc at AOC 617 is F617GW003. Monitoring well F617GW004 is  
31 approximately 50 ft downgradient of the source area.

32 The zinc plume in groundwater is relatively limited in size. Figure 1-5 shows an estimated  
33 area of zinc exceeding the proposed MCS (discussed in Section 2.0 of this report) of 11,000  
34 micrograms per liter ( $\mu\text{g/L}$ ).

#### 1.2.4 Summary of Conclusions and Recommendations from the *RFI Report Addendum and CMS Work Plan (CH2M-Jones, 2001)*

The RFI Report Addendum for AOC 617 concluded the following:

- No surface or subsurface soil COCs were identified.
- Zinc in groundwater within the vicinity of monitoring well F617GW003 was identified as the only groundwater COC.

As a result, the RFI Report Addendum recommended that a focused CMS be undertaken to address zinc in groundwater at AOC 617, within the vicinity of monitoring well F617GW003.

### 1.3 Overall Approach for Selecting Candidate Corrective Measure Alternatives for AOC 617

A variety of corrective measure approaches are conceptually feasible for addressing zinc in groundwater at AOC 617. A Technology Evaluation Report for metals-contaminated soil and groundwater, developed by the Groundwater Remediation Technologies Analysis Center (GWRTAC), describes many of these potentially feasible technologies and is presented in Appendix B of this report. The potentially feasible technologies include:

- Natural attenuation
- In-situ treatment via stabilization/precipitation or electrokinetic processes
- Pump and treat methods, using various aboveground treatment methods
- Permeable reactive barriers (PRBs)

Based on the overall site conditions, CH2M HILL identified the following candidate corrective measure alternatives as the most feasible for the site:

- Natural Attenuation with Land Use Controls (LUCs)
- In-situ stabilization/precipitation
- Pump and treat using relevant extraction and discharge technologies

Electrokinetic methods were considered potentially feasible but likely to require significant study and process evaluation, which would delay implementation of the remedy. Therefore this approach is not considered further in this CMS report. PRBs are considered potentially feasible, but an effective reactive or adsorptive media to remove the zinc from groundwater was not identified during this CMS. Reactive media that are typically used in PRBs (such as iron filings, activated carbon) would not be effective in removing zinc. Although it might be

feasible to identify a suitable media, the time required to do so would require significant study and process testing. Additionally, because of the presence of subsurface utilities in the area and variability in groundwater flow direction due to tidal and potentially seasonal influences, a PRB may not be a suitable remedy for this site.

## 1.4 Report Organization

This CMS report consists of the following sections, including this introductory section:

**1.0 Introduction** — Presents the purpose of and background information relating to this CMS report.

**2.0 Remedial Goal Objectives and Evaluation Criteria**— Defines the RGOs for AOC 617, in addition to the criteria used in evaluating the corrective measure alternatives for the site.

**3.0 Description of Candidate Corrective Measure Alternatives** — Describes each of the candidate corrective measure alternatives for addressing zinc in groundwater.

**4.0 Evaluation and Comparison of Corrective Measure Alternatives** — Evaluates each alternative relative to standard criteria, then compares the alternatives and the degree to which they meet or achieve the evaluation criteria.

**5.0 Recommended Corrective Measure Alternative** — Describes the preferred corrective measure alternative to achieve the MCS and RGOs for zinc in groundwater based on a comparison of the alternatives.

**6.0 References**— Lists the references used in this document.

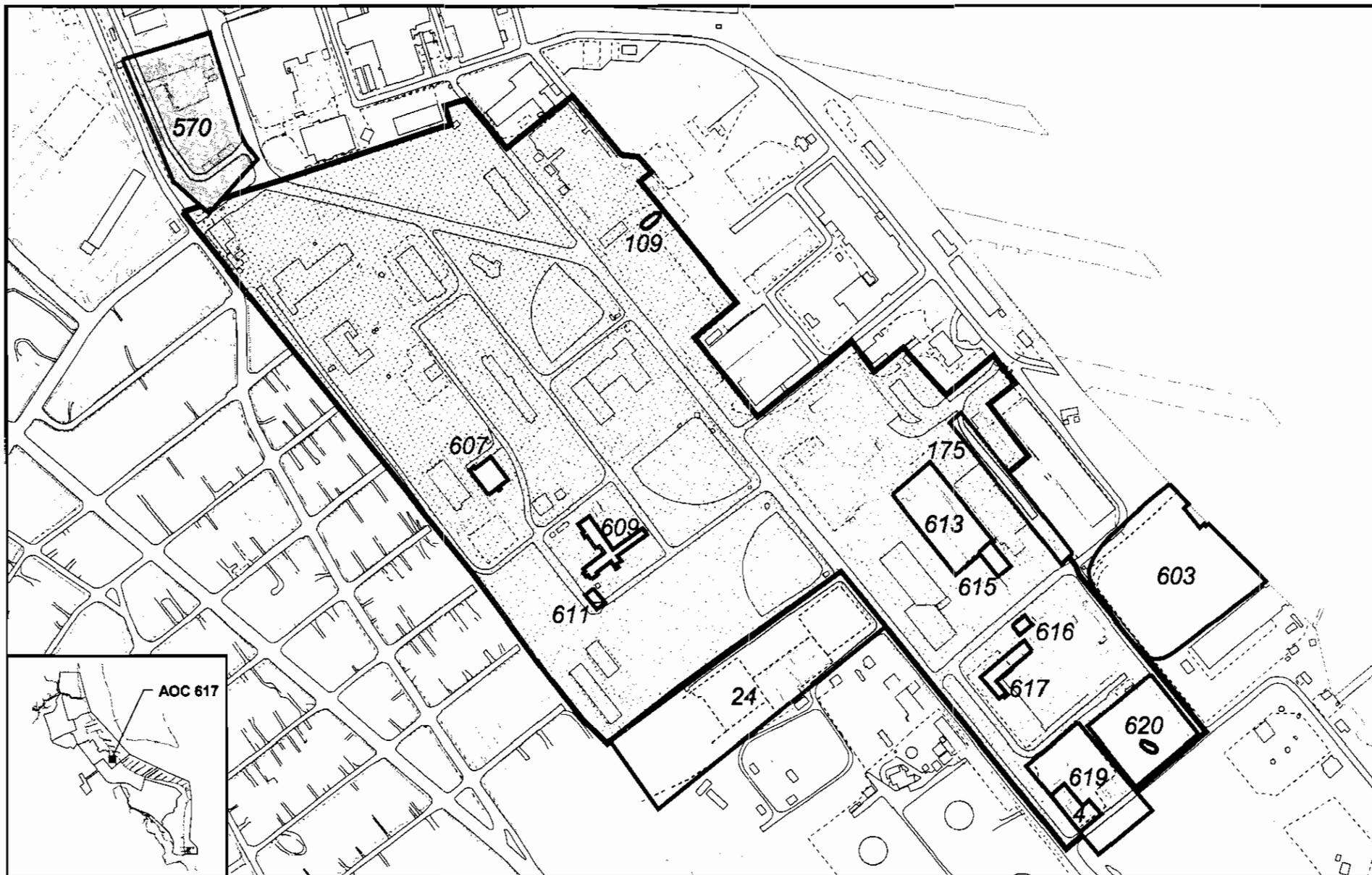
**Appendix A** contains groundwater elevation contours from the *Zone F RFI Report, Revision 0* (EnSafe, 1997).

**Appendix B** contains the Technology Evaluation Report: Remediation of Metals-Contaminated Soils and Groundwater (Ground-Water Remediation Technologies Analysis Center [GWRTAC], 1997).

**Appendix C** contains a summary of Natural Attenuation Modeling (Freeze and Cherry, 1979).

**Appendix D** contains cost estimates developed for the proposed corrective measure alternatives.

All tables and figures appear at the end of their respective sections.

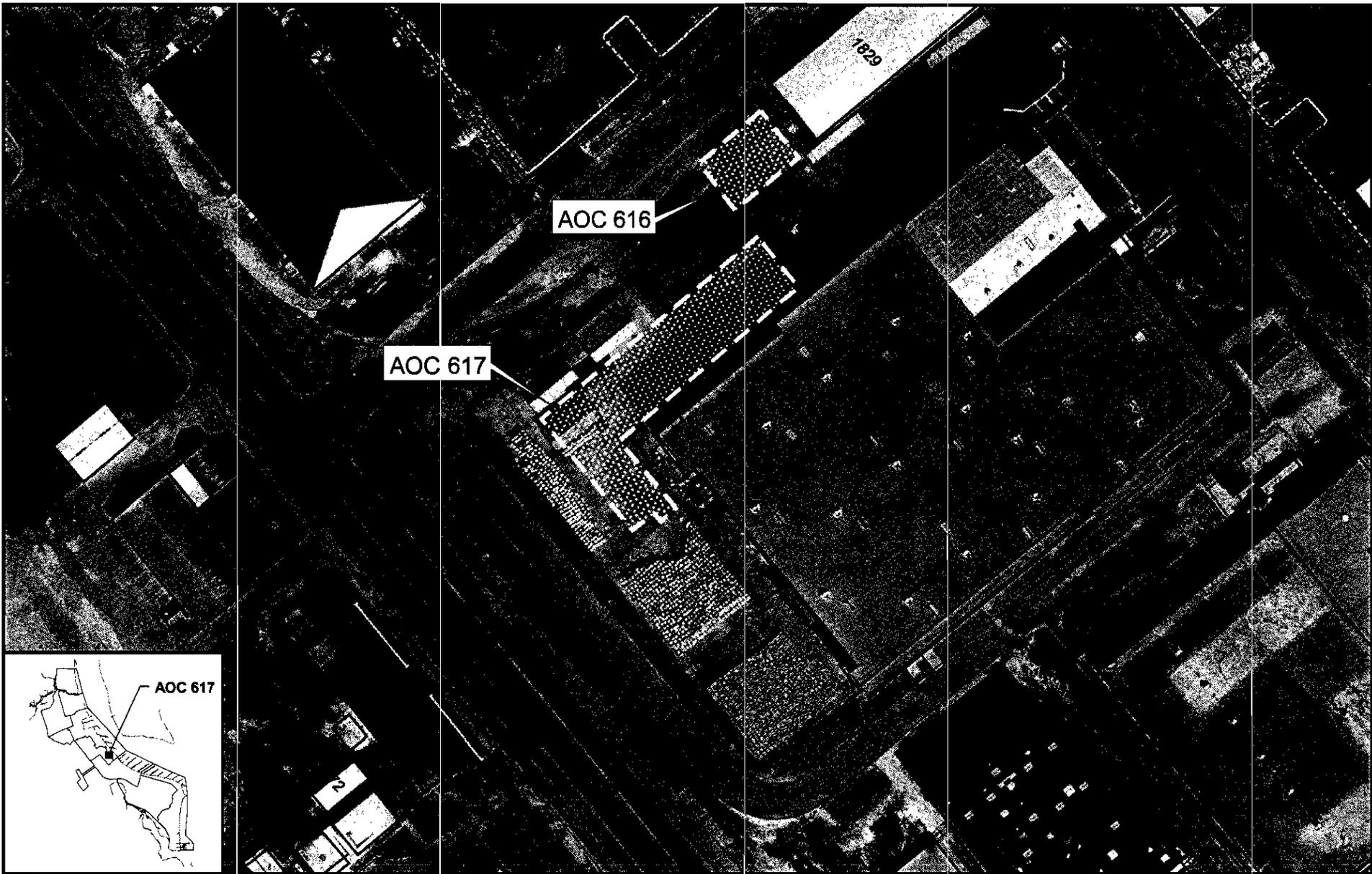


**LEGEND**

-  Zone F Boundary
-  SWMU/AOC Within Zone F Boundary



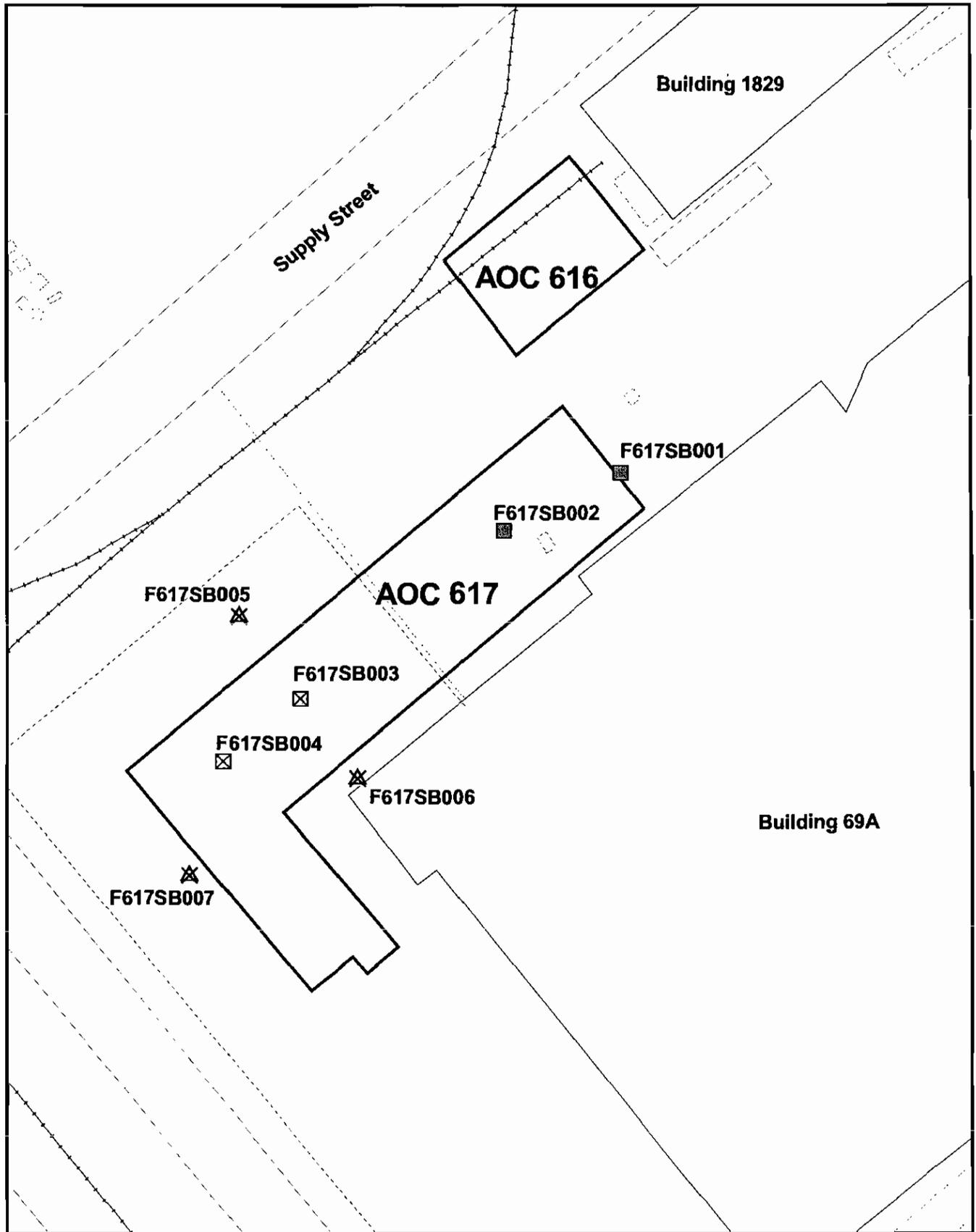
**Figure 1-1**  
 Zone F Within CNC  
 AOC 616 & 617, Zone F  
 Charleston Naval Complex



**LEGEND**  
 [69A] Existing Structure  
 [Dotted Pattern] AOC 616 & 617 Boundary

Note: Aerial Photograph Taken in 1997

**Figure 1-2**  
 Site Layout  
 AOC 616 & 617, Zone F  
 Charleston Naval Complex

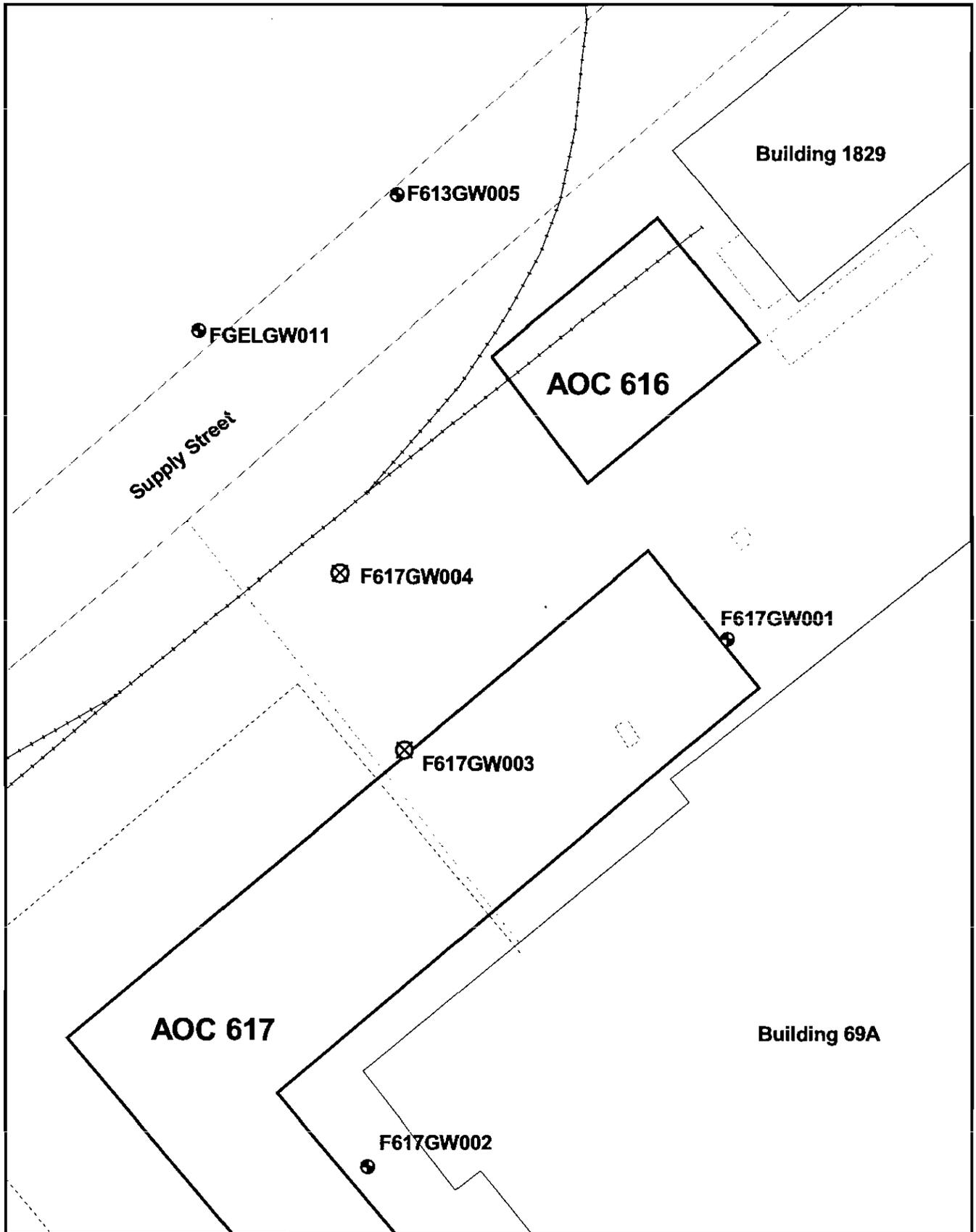


- ★ Soil Sample Locations Installed by CH2M - Jones
- Originally Reported Soil Sample Locations
- ☒ Originally Reported Soil Sample Locations Resampled by EnSafe
- - - Fence
- - - Railroads
- - - Roads
- ▭ AOC Boundary
- ▭ Buildings
- - - Pavement

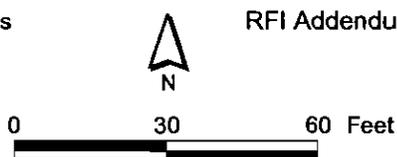


**Figure 1-3**  
RFI Addendum Soil Sample Locations  
AOC 617, Zone F  
Charleston Naval Complex

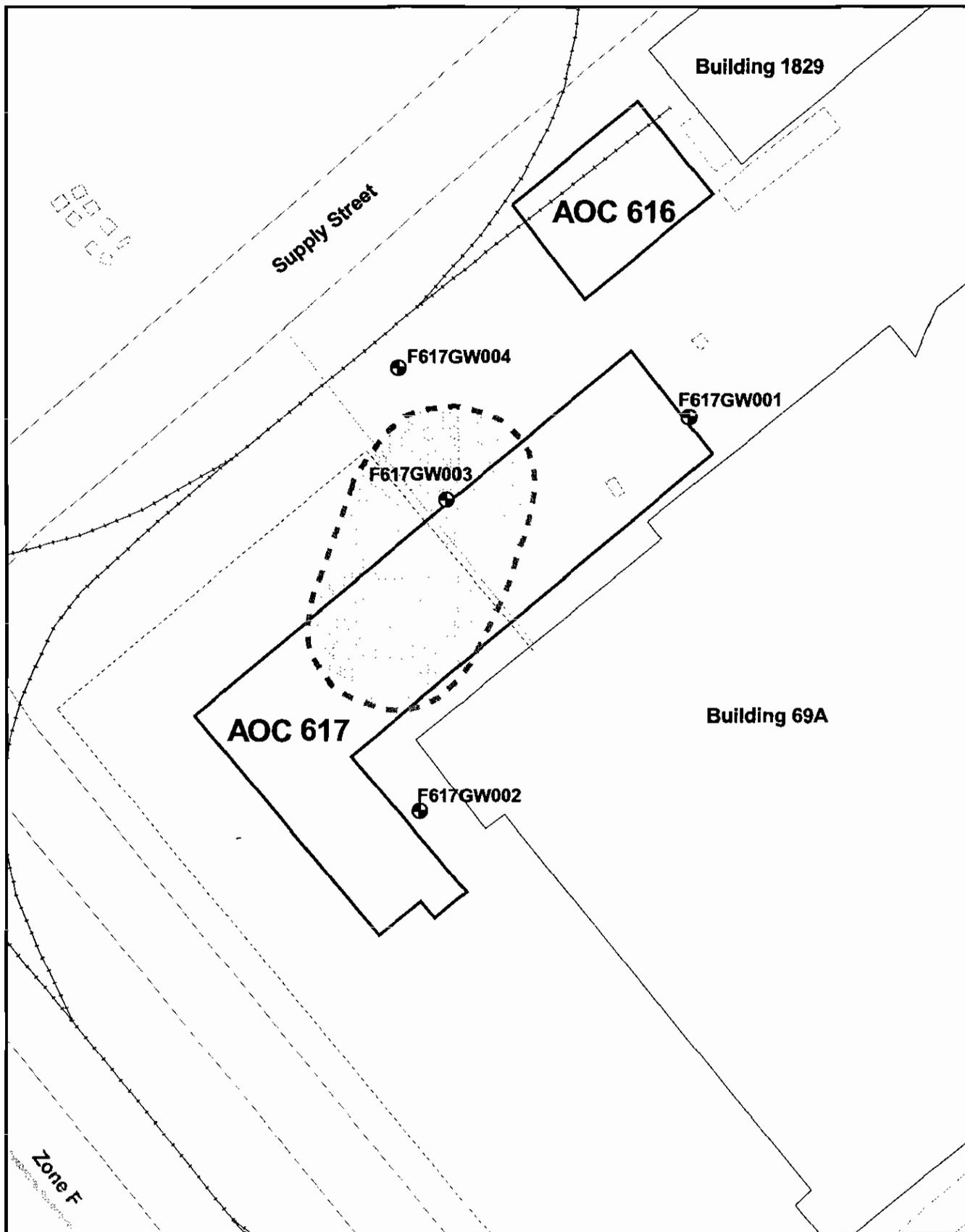




- ⊗ New Groundwater Well Locations
- Originally Reported Groundwater Well Locations
- - - Fence
- - - Railroads
- - - Roads
- ▭ AOC Boundary
- - - Pavement
- ▭ Buildings
- - - Zone Boundary



**Figure 1-4**  
RFI Addendum Groundwater Well Locations  
AOC 616 & 617, Zone F  
Charleston Naval Complex



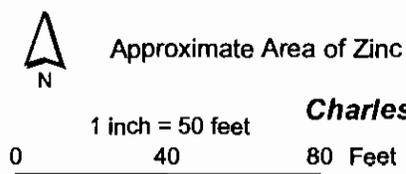
**Figure 1-5**

Approximate Area of Zinc Plume in Groundwater

AOC 617, Zone F

**Charleston Naval Complex**

- Groundwater Monitoring Wells
- ▭ Buildings
- - - Fence
- - - Zone Boundary
- ≡ Railroads
- - - Approximate Area of Zinc Plume
- ≡ Roads
- ▭ AOC Boundary
- ▭ Pavement



Section 2.0

## 2.0 Remedial Goal Objectives and Evaluation Criteria

---

Under RCRA, RGOs and MCSs are typically developed at the end of the risk assessment in the RFI. RGOs can be based on a variety of criteria, such as drinking water maximum contaminant levels (MCLs), specific incremental lifetime cancer risk (ILCR) target levels (e.g., 1E-04, 1E-05, or 1E-06), target Hazard Index (HI) levels (e.g., 0.1, 1.0, 3.0), or site background concentrations. For a particular RGO, specific MCSs can be determined as target concentration values that the selected alternative is required to achieve. Achieving these goals should protect human health and the environment, while achieving compliance with applicable state and federal standards.

### 2.1 Remedial Action Objectives

RAOs are medium-specific goals that protect human health and the environment by preventing or reducing exposures under current and future land use conditions. The RAOs identified for the groundwater at AOC 617 are 1) to prevent ingestion and direct/dermal contact with groundwater having unacceptable non-carcinogenic risk; 2) to prevent migration to offsite areas; and 3) to restore the aquifer to beneficial use. Because no COCs were identified in soils, no RAOs were developed for surface or subsurface soil at AOC 617.

### 2.2 Media Cleanup Standards

RGOs and MCSs for AOC 617 were presented in the RFI Report Addendum and CMS Work Plan (CH2M-Jones, 2001). The focus of this CMS is to evaluate alternatives that will remediate zinc in groundwater at AOC 617. The concentration of zinc in groundwater at the site ranged from 6.37 to 119,000 µg/L during the most recent groundwater sampling event. Since there is no MCL for zinc in groundwater, the MCS/RGO selected is the RBC (11,000 µg/L, based on a HI=1.0). This value is also the EPA Region IX preliminary remediation goal (PRG) for zinc. The groundwater to be addressed is in the vicinity of monitoring well F617GW003.

The corrective measure alternatives to be evaluated include monitored natural attenuation, in-situ stabilization/precipitation, and ex-situ groundwater treatment technologies that allow treated groundwater to be discharged to the sanitary sewer system.

## 2.3 Evaluation Criteria

According to the EPA RCRA CA guidance, corrective measure alternatives should be evaluated using the following five criteria:

1. Protection of human health and the environment
2. Attainment of MCSs
3. The control of the source of releases to minimize future releases that may pose a threat to human health and the environment
4. Compliance with applicable standards for the management of wastes generated by remedial activities
5. Other factors, including (a) long-term reliability and effectiveness; (b) reduction in toxicity, mobility, or volume of wastes; (c) short-term effectiveness; (d) implementability; and (e) cost

Each of these criteria is defined in more detail below:

1. **Protection of human health and the environment.** The alternatives will be evaluated on the basis of their ability to protect human health and the environment. The ability of an alternative to achieve this criterion may or may not be independent of its ability to achieve the other criteria. For example, an alternative may be protective of human health, but may not be able to attain the MCSs if the MCSs were not developed based on human health protection factors.
2. **Attainment of MCSs.** The alternatives will be evaluated on the basis of their ability to achieve the MCS defined in this CMS. Another aspect of this criterion is the time frame required to achieve the MCS. Estimates of the time frame for the alternatives to achieve RGOs will be provided.
3. **The control the source of releases.** This criterion deals with the control of releases of contamination from the source (the area in which the contamination originated) and the prevention of future migration to uncontaminated areas.
4. **Compliance with applicable standards for management of wastes.** This criterion deals with the management of wastes derived from implementing the alternatives (i.e., treatment or disposal of zinc-contaminated residuals from groundwater treatment processes). Corrective measure alternatives will be designed to comply with all standards for management of wastes. Consequently, this criterion will not be explicitly

1 included in the detailed evaluation presented in the CMS, but such compliance would be  
2 incorporated into the cost estimates for which this criterion is relevant.

3 5. **Other factors.** Five other factors are to be considered if an alternative is found to meet  
4 the four criteria described above. These other factors are as follows:

5 a. Long-term reliability and effectiveness

6 Corrective measure alternatives will be evaluated on the basis of their reliability, and  
7 the potential impact should the alternative fail. In other words, a qualitative  
8 assessment will be made as to the chance of the alternative's failing and the  
9 consequences of that failure.

10 b. Reduction in the toxicity, mobility, or volume of wastes

11 Alternatives with technologies that reduce the toxicity, mobility, or volume of the  
12 contamination will be generally favored over those that do not. Consequently, a  
13 qualitative assessment of this factor will be performed for each alternative.

14 c. Short-term effectiveness

15 Alternatives will be evaluated on the basis of the risk they create during the  
16 implementation of the remedy. Factors that may be considered include fire,  
17 explosion, and exposure of workers to hazardous substances.

18 d. Implementability

19 The alternatives will be evaluated for their implementability by considering any  
20 difficulties associated with conducting the alternatives (such as the construction  
21 disturbances they may create), operation of the alternatives, and the availability of  
22 equipment and resources to implement the technologies comprising the alternatives.

23 e. Cost

24 A net present value of each alternative will be developed. These cost estimates will  
25 be used for the relative evaluation of the alternatives, not to bid or budget the work.  
26 The estimates will be based on information available at the time of the CMS and on a  
27 conceptual design of the alternative. They will be "order-of-magnitude" estimates  
28 with a generally expected accuracy of -50 percent to +100 percent for the scope of  
29 action described for each alternative. The estimates will be categorized into capital  
30 costs and operations and maintenance costs for each alternative.

Section 3.0

## 1 **3.0 Description of Candidate Corrective** 2 **Measure Alternatives**

---

### 3 **3.1 Introduction**

4 Three candidate corrective measure alternatives were selected for this site:

- 5 • Alternative 1: Natural Attenuation with LUCs
- 6 • Alternative 2: In-situ Stabilization/Precipitation
- 7 • Alternative 3: Groundwater Extraction, Treatment, and Discharge to Sanitary Sewer

8 The sections below describe each alternative in detail.

### 9 **3.2 Alternative 1: Natural Attenuation with Land Use Controls**

#### 10 **3.2.1 Description of Alternative**

11 This alternative will allow the zinc to naturally attenuate in the subsurface, will impose  
12 LUCs (such as a deed restriction) to restrict the installation of drinking water wells, and will  
13 monitor groundwater concentrations periodically until the MCS is reached.

14 Information on groundwater flow indicates that the groundwater is flowing to the north-  
15 northeast into a groundwater "trough" that trends east-west. Elevated concentrations of  
16 zinc that were detected at F617GW002 in 1997 appear to have migrated downgradient and  
17 are now in the area located near F617GW003. The groundwater migration rate in this area is  
18 approximately 0.1 ft/day. As noted in the GWRTAC report, which is provided in Appendix  
19 B, zinc is one of the most mobile heavy metals in groundwater at acidic and neutral pHs. As  
20 a conservative assumption, the maximum potential migration rate of zinc could be assumed  
21 to be close to the groundwater advection rate. Downgradient of F617GW003, it appears that  
22 the direction of groundwater flow shifts towards the north-northwest, and the rate of  
23 migration in this area is expected to be significantly less since the gradient is approximately  
24 4 times lower. This would correspond to a groundwater velocity of approximately .03  
25 ft/day.

### 1 **3.2.2 Key Uncertainties**

2 A key uncertainty for the natural attenuation alternative is whether the zinc groundwater  
3 plume would discharge to a nearby water body (such as the Cooper River) by either direct  
4 discharge or via interception into a leaking storm sewer before the zinc had attenuated to  
5 concentrations that would not cause an unacceptable impact or risk to the environment.

6 Another uncertainty is whether the zinc plume might be intercepted by a leaking sanitary  
7 sewer at concentrations above the permitted sewer discharge standards. Adequate  
8 monitoring of the zinc plume would be necessary to make sure that these scenarios are  
9 avoided.

10 In order to assess on a preliminary basis the attenuation and migration characteristics of the  
11 zinc plume, an analytical solution described in Freeze and Cherry (1979) was used to predict  
12 the amount of time needed to reduce the concentration in the zinc source area to less than  
13 the MCS. The details of this evaluation are presented in Appendix C. The solution assumes  
14 that the source of contamination is a slug (i.e., there is no ongoing source of contamination),  
15 and that there is no adsorption of the dissolved contaminant onto the aquifer matrix. The  
16 solution results are dependent upon several parameters, including initial mass of  
17 contaminant, porosity, groundwater velocity, and dispersivity.

18 Using estimates for these parameters that are based on a combination of site data and  
19 contaminant transport literature, the analytical solution indicates that the zinc plume would  
20 attenuate in approximately 30 years. During that time, the plume would migrate (in a  
21 theoretical homogenous, isotropic aquifer) a distance of between 1,000 and 4,000 feet. It is  
22 important to note that if retardation is a significant attenuating mechanism for zinc, the  
23 estimates of attenuation time and migration distance could be considerably less. However,  
24 this result also suggests that the plume could be potentially intercepted by one of the  
25 various storm and sanitary sewers or even discharge to the Cooper River before being  
26 completely attenuated, since there are a number of sewers within 1,000 feet downgradient of  
27 the site, and the Cooper River is only 760 feet from the site.

### 28 **3.2.3 Other Considerations**

29 LUCs would be necessary to prevent installation of drinking water wells at AOC 617 until  
30 adequate attenuation of zinc had occurred. Periodic groundwater monitoring would also be  
31 necessary to ensure that unacceptable impacts to receptors are not occurring.

1 Based on the above considerations, there appear to be significant uncertainties that would  
2 need to be resolved to better understand the viability of this approach prior to its  
3 implementation.

## 4 **3.3 Alternative 2: In-situ Stabilization/Precipitation**

### 5 **3.3.1 Description of Alternative**

6 This alternative involves the injection of a stabilization or precipitation agent, such as a  
7 sulfide- or hydroxide-based material, to precipitate the zinc from the dissolved phase and  
8 into a solid phase. The precipitating material could be delivered to the aquifer via a variety  
9 of methods, including liquid or gas injection. Process and design parameters would need to  
10 be determined through the performance of bench-scale and most likely pilot-scale testing,  
11 before the feasibility of the approach is fully known.

### 12 **3.3.2 Key Uncertainties**

13 The greatest uncertainty is the long-term stability of the zinc precipitate. A process using a  
14 sulfide system may be sensitive to long-term changes in oxidation-reduction potential (ORP)  
15 in the groundwater. As long as the site stays under reducing conditions, the zinc would  
16 likely stay stable as a sulfide precipitate. If ORP increases, some conversion of the sulfide to  
17 sulfate is feasible, which may release some zinc into solution. Similarly, for a precipitation  
18 process based on hydroxide, a decrease in groundwater pH could result in a release of  
19 precipitated zinc back into the dissolved state.

20 It is also uncertain whether periodic injections of precipitating reagents might be needed to  
21 maintain the zinc concentrations below the MCS. In addition, the ideal precipitating agent  
22 and related chemical conditions, as well as the effectiveness of specific potential injection  
23 methods to deliver the reagents to the necessary areas, are unknown.

### 24 **3.3.3 Other Considerations**

25 Periodic monitoring of the groundwater zinc concentrations, pH, ORP, and other chemical  
26 parameters would be essential for measuring the effectiveness of this alternative. For the  
27 purpose of developing a representative cost estimate for this process, a precipitation process  
28 based on a lime slurry injection was assumed.

29 Based on the above considerations, there appear to be significant uncertainties that would  
30 need to be resolved to better understand the viability of this approach prior to its  
31 implementation.

## 3.4 Alternative 3: Groundwater Extraction, Treatment, and Discharge to the Sanitary Sewer

### 3.4.1 Description of Alternative

This alternative is basically a pump and treat approach, in which zinc-contaminated groundwater is recovered and treated to allow discharge to the sanitary sewer system. Depending on the zinc concentration in recovered groundwater, treatment prior to discharge to the sewer may or may not be necessary. The sanitary sewer authority in this area is Charleston Public Works (CPW). The current zinc discharge standard for the CPW sanitary sewer is 5,000 µg/L. If treatment prior to discharge is required, a variety of treatment processes are conceptually available, including chemical (alkaline) precipitation, ion exchange, and reverse osmosis. The key elements of this approach include:

1. A groundwater recovery system, including operational controls and conveyance piping
2. Aboveground treatment for zinc removal
3. A discharge line to the sanitary sewer

For groundwater recovery, a variety of approaches are feasible, including vertical recovery wells, horizontal wells, or recovery trenches. Given the localized nature of the zinc plume, vertical recovery wells appear to be the most appropriate. Conceptually, one or several recovery wells would be located slightly downgradient of monitoring well F617GW003. These wells would recover contaminated groundwater and preclude further downgradient migration of the zinc plume.

For estimating the potential cost for this type of system, CH2M Jones assumed that aboveground treatment of the recovered groundwater would be performed using an alkaline precipitation via hydroxide. The zinc in groundwater would combine with hydroxide to form insoluble zinc hydroxide. Periodic removal and disposal of the zinc precipitate would be required. The precipitate would be dewatered (e.g., using a plate and frame press or similar methods), and the solids would be disposed of off site. It is expected that the waste sludge would be non-hazardous.

### 3.4.2 Key Uncertainties

The key uncertainties for this alternative include:

1. The long-term sustainable groundwater recovery rate

1 2. The concentration of zinc in the recovered groundwater and length of time that its  
2 concentration would exceed the sanitary sewer discharge standard for zinc

3 3. The length of time required to recover groundwater to achieve the MCS

4 Regarding the first uncertainty, the sustainable groundwater recovery rate, it is expected  
5 that the rate is relatively low, less than 5 gallons per minute (gpm). The best way to  
6 determine the long-term sustainable rate prior to implementing this alternative would be to  
7 conduct a 48- to 72- hour pumping test. Water level drawdowns in the test pumping well, as  
8 well as in nearby observation wells, would be monitored and recorded, then used to  
9 determine site-specific aquifer characteristics. These data would allow for the long-term  
10 sustainable aquifer yield to be estimated.

11 Regarding the concentration of zinc in the recovered groundwater, experience at many  
12 pump and treat systems has shown that the concentrations of target contaminants in  
13 groundwater collected from recovery wells are nearly always lower, often significantly  
14 lower, than concentrations measured in monitoring wells under static groundwater  
15 conditions. Additionally, experience has shown that concentrations of target contaminants  
16 in groundwater from recovery wells decrease over time, as the plume is cleaned up.  
17 Typically, the decrease in concentration of the contaminant in recovered groundwater over  
18 time follows an exponentially decreasing curve. Conducting a pump test would also  
19 provide an opportunity to measure the concentration of zinc in the recovered groundwater  
20 and assess any changes in concentration over the test period.

21 In order to provide a preliminary assessment of the potential duration of required  
22 groundwater recovery and treatment, an estimate of the total volume of contaminated  
23 groundwater within the plume was developed. The volume of groundwater that requires  
24 treatment is estimated to be approximately 500,000 to 700,000 gallons, based on a  
25 contaminated aquifer volume that measures 150 ft by 100 ft by 15 ft-thick, and a porosity  
26 range of 30 to 40 percent. If the average extraction rate is 2 gpm, approximately 0.5 to 0.7  
27 years would be required to extract one pore volume. Given that much of the zinc is expected  
28 to be in a highly dissolved state, the recovery of the first several pore volumes may result in  
29 the removal of most of the dissolved groundwater plume that requires treatment.

30 Zinc may continue to desorb from aquifer solids after removal of the several pore volumes  
31 at a rate that requires continued pumping and treating. However, if the residual adsorbed  
32 zinc mass and its rate of desorption is limited, the zinc concentration of the recovered  
33 groundwater may be low enough such that it could be discharged to the sanitary sewer

1 without requiring treatment, or that monitored natural attenuation may become a more  
2 viable alternative.

### 3 **3.4.3 Other Considerations**

4 LUCs to prevent the installation of drinking water wells at AOC 617 would be required  
5 during the timeframe when groundwater zinc concentrations are greater than the MCS.  
6 Periodic monitoring of groundwater wells and the system performance would be required  
7 to ensure that it is operating as intended.

8 Also, it is possible that after a pump and treat approach has removed several pore volumes  
9 of the zinc-contaminated groundwater from the aquifer, the viability of natural attenuation  
10 or the use of an in-situ precipitation process may be more viable and could be reconsidered  
11 as a feasible remedy at a later time.

Section 4.0

## 4.0 Evaluation and Comparison of Corrective Measure Alternatives

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The three corrective measure alternatives were evaluated relative to the evaluative criteria previously described in Section 2.0, and then subjected to a comparative evaluation. A cost estimate for each alternative was also developed; the assumptions and unit costs used for these estimates are included in Appendix C.

### 4.1 Alternative 1: Natural Attenuation with Land Use Controls

The assumptions for Alternative 1 include the following:

- A base-wide land use control management plan (LUCMP) will be developed for the CNC. The plan will allow for restrictions on the use of groundwater at AOC 617 and other areas, and will be developed outside the scope of this CMS.
- Periodic groundwater monitoring will be performed for 20 years. Samples will be collected from up to 10 groundwater wells, on at least a semi-annual basis.

#### 4.1.1 Protection of Human Health and the Environment

This alternative is effective at protecting human health because it uses LUCs to prevent the ingestion of and direct contact with groundwater. With regard to protection of the environment, monitoring would need to be conducted to ensure that the zinc plume does not migrate into the Cooper River via direct discharge or by interception by a storm sewer, such that it could create unacceptable environmental impacts. If so, additional, active corrective measures would need to be implemented to preclude such impacts.

#### 4.1.2 Attain MCS

This alternative is expected to eventually attain the MCS (in approximately 30 years).

#### 4.1.3 Control the Source of Releases

There are no ongoing sources of releases at AOC 617. However, this alternative could allow the migration of zinc into downgradient, uncontaminated groundwater.

1 **4.1.4 Compliance with Applicable Standards for the Management of Generated**  
2 **Wastes**

3 Alternative 1 does not generate any wastes that require special management. The primary  
4 generated waste would be purge water from monitoring wells, which is easily managed to  
5 applicable standards.

6 **4.1.5 Other Factors (a) Long-term Reliability and Effectiveness**

7 This alternative has adequate long-term reliability and effectiveness, provided that  
8 migration of the plume at unacceptable concentrations into surface water or the sanitary  
9 sewer does not occur. If such migration occurred, additional corrective measures would  
10 likely be necessary.

11 **4.1.6 Other Factors (b) Reduction in the Toxicity, Mobility, or Volume of Wastes**

12 Alternative 1 relies on natural attenuation to reduce the toxicity of the contaminated  
13 groundwater. This alternative does not reduce the mobility or volume of contaminated  
14 groundwater.

15 **4.1.7 Other Factors (c) Short-term Effectiveness**

16 Through the implementation of LUCs, this alternative has short-term effectiveness in  
17 preventing ingestion of or contact with the contaminated groundwater. No significant short-  
18 term risks would be created using this alternative.

19 **4.1.8 Other Factors (d) Implementability**

20 Alternative 1 is easily implemented since it only requires the implementation of LUCs and  
21 an appropriate monitoring well program.

22 **4.1.9 Other Factors (e) Cost**

23 Alternative 1 is the least costly to implement since it requires no construction of treatment  
24 facilities or disposal of wastes. The significant component of cost for this alternative is for  
25 groundwater monitoring.

26 Using the assumptions described earlier, the total present value of this alternative is  
27 \$256,000.

28 **4.2 Alternative 2: In-situ Stabilization/Precipitation**

29 A presumptive approach of using a lime (hydroxide-based) precipitation process was  
30 assumed for evaluating this alternative. The following other assumptions were made:

- 1 • Quarterly groundwater monitoring would be performed at eight wells for a duration of
- 2 5 years.
- 3 • Semi-annual groundwater monitoring would be performed at eight wells for a
- 4 subsequent duration of 15 years.
- 5 • A yearly cost was included for the first five years for the injection of additional lime to
- 6 better optimize zinc precipitation.

#### 7 **4.2.1 Protection of Human Health and the Environment**

8 This alternative is effective at protecting human health and the environment because it uses  
9 LUCs to prevent the ingestion of and direct contact with groundwater during the time  
10 period when groundwater zinc concentrations are greater than the MCS.

#### 11 **4.2.2 Attain MCS**

12 It is unclear whether this alternative will be able to permanently achieve the MCS.  
13 Additional injection of lime slurry may be needed if subsurface conditions (such as pH)  
14 change and cause the zinc to resolubilize. Using an effective precipitation process, the MCS  
15 could likely be achieved within one year after implementation.

#### 16 **4.2.3 Control the Source of Releases**

17 There are no ongoing sources of releases at AOC 617, therefore this issue is not applicable.  
18 This alternative would immobilize the zinc, precluding downgradient migration into  
19 uncontaminated groundwater.

#### 20 **4.2.4 Compliance with Applicable Standards for the Management of Generated** 21 **Wastes**

22 This alternative does not generate any wastes that require special management.

#### 23 **4.2.5 Other Factors (a) Long-term Reliability and Effectiveness**

24 This alternative has long-term reliability because of the implementation of LUCs.  
25 Groundwater concentrations may rebound as zinc that may be adsorbed to the aquifer  
26 matrix slowly partitions into the groundwater. This may result in having to re-implement  
27 Alternative 2 some time period after the first injection.

#### 28 **4.2.6 Other Factors (b) Reduction in the Toxicity, Mobility, or Volume of Wastes**

29 Alternative 2 reduces the toxicity, mobility, and volume of the contaminated groundwater.

#### 1 **4.2.7 Other Factors (c) Short-term Effectiveness**

2 Because of the implementation of LUCs, this alternative will have short-term effectiveness in  
3 preventing ingestion of or contact with the contaminated groundwater. Because the  
4 precipitation reaction is relatively rapid, this alternative would have short-term  
5 effectiveness in precipitating the zinc into the solid phase. No unmanageable hazards would  
6 be created during its implementation.

#### 7 **4.2.8 Other Factors (d) Implementability**

8 This alternative may be moderately difficult to implement because of the problems inherent  
9 to the subsurface injection of the lime slurry, but could be performed without excessive  
10 difficulty.

#### 11 **4.2.9 Other Factors (e) Cost**

12 A cost estimate was provided by ARS Technologies for the injection of lime slurry.  
13 Appendix C presents the overall cost estimate for implementing this remedy. A pilot test,  
14 prior to the design of the system, is also included in this cost estimate.

15 Using the cost estimate provided by ARS Technologies and the assumptions listed above,  
16 the total present value of the alternative is \$790,000. This cost estimate assumes that  
17 repeated injections of lime-slurry will be necessary to maintain proper subsurface conditions  
18 for the first five years.

### 19 **4.3 Alternative 3: Groundwater Extraction, Treatment and** 20 **Discharge to Sanitary Sewer**

21 The assumptions for developing a conceptual cost estimate for Alternative 3 include the  
22 following:

- 23 • Groundwater extraction will occur via two recovery wells located slightly downgradient  
24 of monitoring well F617GW003.
- 25 • Piping will be polyvinyl chloride (PVC) and located underground to minimize  
26 disruption of activities at Building 69A.
- 27 • Hydroxide precipitation will be used as the treatment method. Two tanks would be  
28 used in the treatment system. The first tank will hold the water prior to treatment, and  
29 the second tank will hold the water after treatment for chemical analysis prior to  
30 discharge to the sewer.

- 1 • Sludge generated by the process is non-hazardous.

2 For the purpose of preparing a conservative cost estimate, the treatment system is assumed  
3 to operate for five years. After this, continued pumping will be required for three additional  
4 years but the recovered groundwater could be discharged to the sewer without treatment.

#### 5 **4.3.1 Protection of Human Health and the Environment**

6 This alternative is effective at protecting human health because it uses LUCs to prevent the  
7 ingestion of and direct contact with groundwater during the time period when groundwater  
8 zinc concentrations are greater than the MCS. It protects the environment by preventing  
9 migration of the zinc plume into nearby surface water.

#### 10 **4.3.2 Attain MCS**

11 It is believed that Alternative 3 may attain the MCS within 10 years, possibly in less than  
12 five years, after pumping is initiated. The timeframe is uncertain because of uncertainty in  
13 the volume estimate for the contaminated groundwater, and the amount of zinc that may be  
14 adsorbed onto the aquifer matrix, as well as the other factors previously discussed.

#### 15 **4.3.3 Control the Source of Releases**

16 There are no ongoing sources of releases at AOC 617, therefore this issue is not applicable.  
17 This alternative would prevent the migration of zinc into downgradient, uncontaminated  
18 groundwater.

#### 19 **4.3.4 Compliance with Applicable Standards for the Management of Generated 20 Wastes**

21 Alternative 3 generates a waste stream (zinc hydroxide sludge) that will require disposal.  
22 The handling, dewatering, and disposal of this waste has the potential to have a significant  
23 impact on cost; but it is assumed that this waste would be non-hazardous and that  
24 compliance with applicable disposal standards would not present a significant issue.

#### 25 **4.3.5 Other Factors (a) Long-term Reliability and Effectiveness**

26 Alternative 3 has long-term reliability because of the implementation of LUCs and active  
27 removal of the zinc from the groundwater.

#### 28 **4.3.6 Other Factors (b) Reduction in the Toxicity, Mobility, or Volume of Wastes**

29 This alternative reduces toxicity, mobility, and volume of contaminated environmental  
30 media by removing the zinc from the aquifer.

1 **4.3.7 Other Factors (c) Short-term Effectiveness**

2 Because of the implementation of LUCs, this alternative has short-term effectiveness in  
3 preventing ingestion of or contact with the contaminated groundwater. Because of the active  
4 recovery of groundwater, potential threats to the environment are quickly minimized. No  
5 unmanageable hazards would be created during its implementation.

6 **4.3.8 Other Factors (d) Implementability**

7 Alternative 3 is relatively easy to implement since the system is small and relies on  
8 conventional technologies.

9 **4.3.9 Other Factors (e) Cost**

10 The total present value for this alternative as it is conceptually envisioned is \$410,000,  
11 placing Alternative 3 in the middle of the cost range. One of the assumptions made in the  
12 cost estimate is that the zinc precipitate can be disposed of at a non-hazardous landfill. If the  
13 zinc precipitate is not inert, as it is expected to be, it may require special disposal  
14 considerations which will add to the cost.

15 **4.4 Comparative Ranking of Corrective Measure Alternatives**

16 Each corrective measure alternative's overall ability to meet the evaluation criteria is  
17 described above. In the table below, a comparative evaluation of the degree to which each  
18 alternative meets a particular criteria is presented. For all of the criteria except cost, an  
19 alternative was given a score of 1 if it did not meet a criterion, 3 if it moderately met the  
20 criterion, and 5 if it fully met the criterion. Alternatives with low, moderate, and high  
21 relative estimated costs were given a score of 5, 3, and 1 respectively. The scores for each  
22 alternative were summed to give an overall score, as shown in Table 4-1. The higher the  
23 total score, the better that alternative met the evaluation criteria.

**TABLE 4-1**  
 Ranking of Corrective Measure Alternatives  
*Corrective Measures Study Report, AOC 617, Zone F, Charleston Naval Complex*

<b>Criterion</b>	<b>1. Monitored Natural Attenuation</b>	<b>2. In-Situ precipitation</b>	<b>3. Groundwater Extraction, Treatment and Discharge</b>
Overall Protection of Human Health and the Environment	3	5	5
Attainment of MCS	1	3	5
Control of the source of releases	1	5	5
Compliance with applicable standards for the management of wastes	5	5	5
Long-term Reliability and Effectiveness	2	3	5
Reduction of Toxicity, Mobility, or Volume through Treatment	1	3	5
Short-term Effectiveness	2	4	4
Implementability	5	1	3
Cost Ranking	5	1	3
Estimated Cost (in \$1,000)	\$256	\$790	\$410
<b>Overall Score</b>	<b>25</b>	<b>30</b>	<b>40</b>

Section 5.0

## 5.0 Recommended Corrective Measure Alternative

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Three corrective measure alternatives were evaluated using the criteria described in Section 2.0 of this CMS report. These alternatives included: Alternative 1: Natural Attenuation with LUCs; Alternative 2: In-Situ Stabilization/Precipitation; and Alternative 3: Groundwater Extraction, Treatment, and Discharge to the Sanitary Sewer. The RAOs identified for groundwater at AOC 617 are: 1) to prevent ingestion and direct/dermal contact with groundwater having unacceptable carcinogenic or noncarcinogenic risk; 2) to prevent migration to offsite areas; and 3) to restore the aquifer to beneficial use.

Based on the alternatives evaluation and RAOs for the site and current uncertainties associated with each alternative, the preferred corrective measure alternative is Alternative 3: Groundwater Extraction, Treatment, and Discharge to the Sanitary Sewer. The RAO of preventing ingestion and direct/dermal contact with contaminated groundwater is achieved at a moderate cost. The second RAO exists because of potential ingestion or contact that could occur during intrusive site maintenance or if the plume migrates off site; the implementation of groundwater extraction and treatment will ensure that plume migration will not occur because of active pumping and because the mass of zinc in the groundwater will be reduced over time. The final RAO of restoring the aquifer to beneficial use will be met when the zinc concentrations in the aquifer are less than or equal to the MCS.

As discussed in Section 3.0, each of the alternatives has significant uncertainty. In order to better assess the viability and appropriateness of Alternative 3 as a long-term remedy, it is recommended that the horizontal and vertical extent of groundwater exceeding the MCS of 11,000 µg/L first be better delineated (via groundwater profiling investigation of the area). After the dimensions of the target treatment area have been established, a 48- to 72-hour pump test should be implemented in the vicinity of F617GW003. The pump test would provide necessary information for the following:

- to allow the determination of the long-term sustainable groundwater recovery rate;
- to observe any short-term changes in zinc concentrations and pH in recovered groundwater; and

- 1 • provide design information that would allow the design of the groundwater recovery  
2 and treatment system.

3 A work plan for this groundwater investigation and pump test should be developed that  
4 would specify the design and locations of the groundwater samples to be analyzed as well  
5 as locations and design of test recovery well and observation wells to be used, the approach  
6 for monitoring well drawdowns and evaluating the drawdown data, the sampling and  
7 analysis regime to be followed, and approach to handling recovered groundwater and other  
8 related wastes (such as drill cuttings and purge water).

9 After completion of the groundwater assessment and subsequent pump test, a revised  
10 conceptual configuration for a pump and treat system should be developed and a  
11 determination made as to whether Alternative 3 continues to be the preferred alternative.

12 The remedial design would then be completed and the remedy implemented.

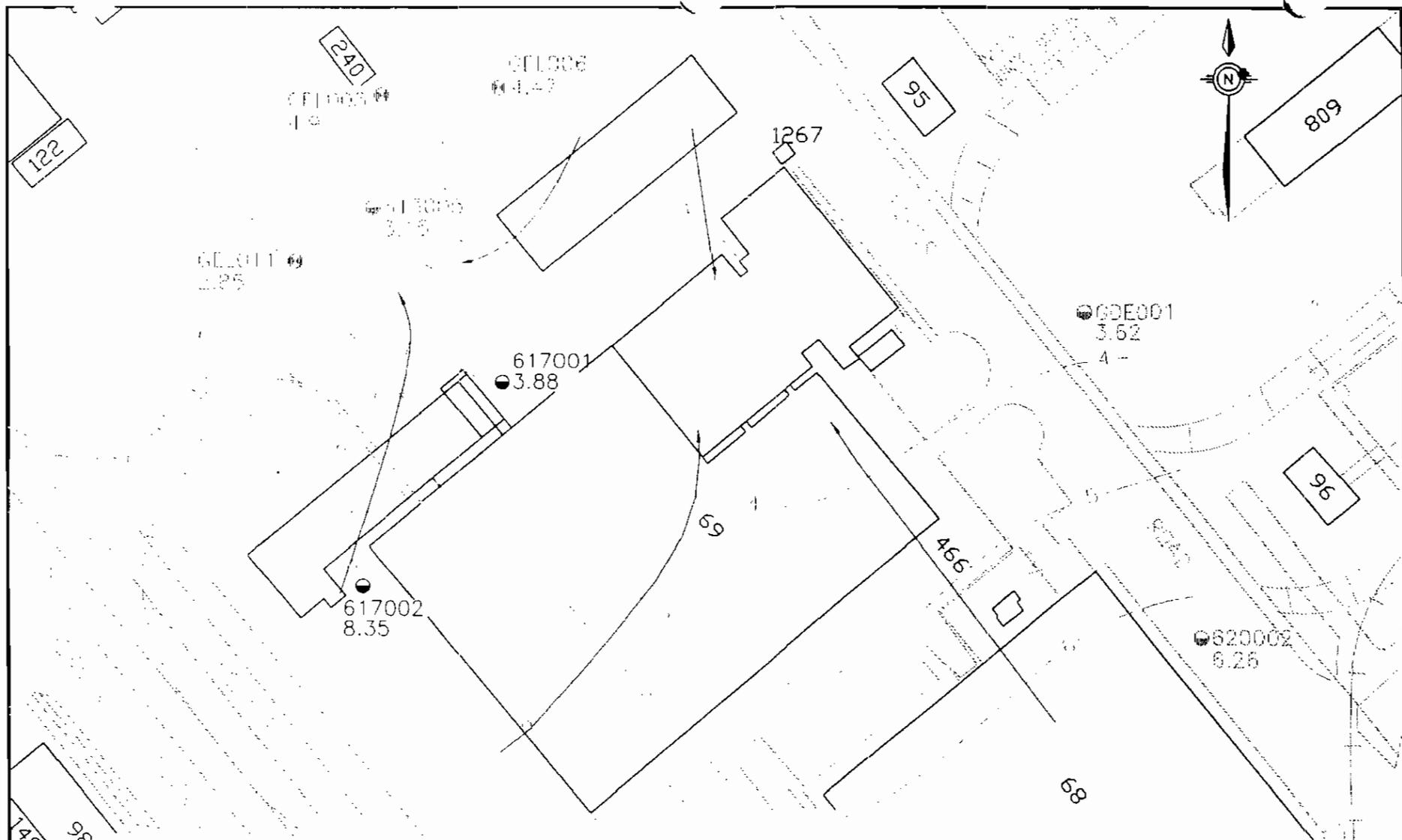


## 1 **6.0 References**

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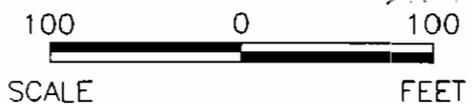
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- 9 South Carolina Department of Health and Environmental Control (SCDHEC). *RCRA Permit*  
10 *SC0 170 022 560. Charleston Naval Complex, Charleston, South Carolina. August 17, 1988.*
- 11 South Carolina Department of Health and Environmental Control (SCDHEC). *Comments on*  
12 *Zone F RFI Report, Revision 0. December 31, 1998.*

Appendix A



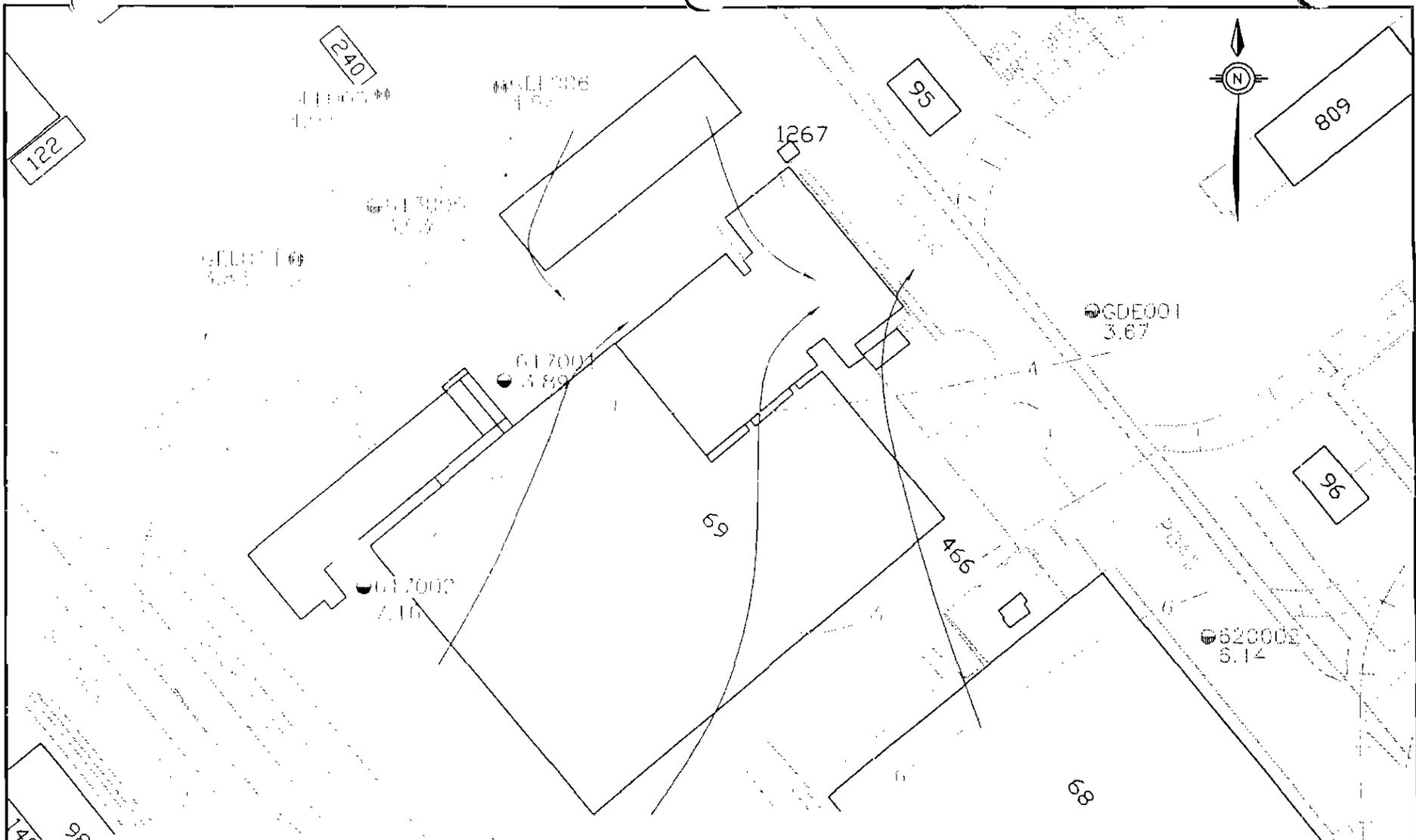
**LEGEND**

- - SHALLOW MONITORING WELL
- ⊙ - ADJACENT SHALLOW MONITORING WELL
- ⊕ - ADJACENT SHALLOW MONITORING WELL NOT INSTALLED BY ENSAFE
- - - - - CONTOUR INTERVAL - 1 FOOT
- - FLOW DIRECTION



ZONE F  
 RCRA FACILITY  
 INVESTIGATION REPORT  
 NAVAL BASE CHARLESTON  
 CHARLESTON, S.C.

FIGURE 10.9-2  
 SHALLOW GROUNDWATER  
 LOW-TIDE POTENTIOMETRIC MAP  
 AOC #617, GALVANIZING PLANT  
 FORMER BLDG 1176  
 DWG DATE: 10/30/97 | DWG NAME: 2906GPLT



**LEGEND**

- - SHALLOW MONITORING WELL
- - ADJACENT SHALLOW MONITORING WELL
- - ADJACENT SHALLOW MONITORING WELL NOT INSTALLED BY ENSAFE
- - - - - CONTOUR INTERVAL - 1 FOOT
- - - - - FLOW DIRECTION



ZONE F  
 RCRA FACILITY  
 INVESTIGATION REPORT  
 NAVAL BASE CHARLESTON  
 CHARLESTON, S.C.

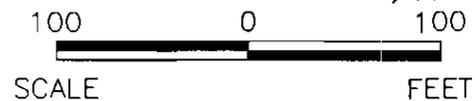


FIGURE 10.9-3  
 SHALLOW GROUNDWATER  
 HIGH-TIDE POTENTIOMETRIC MAP  
 AOC #617, GALVANIZING PLANT  
 FORMER BLDG 1176  
 DWG DATE: 10/30/97 | DWG NAME: 2906GPHT

Appendix B

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# Remediation of Metals-Contaminated Soils and Groundwater

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## FOREWORD

### About GWRTAC

The Ground-Water Remediation Technologies Analysis Center (GWRTAC) is a national environmental technology transfer center that provides information on the use of innovative technologies to clean-up contaminated groundwater.

Established in 1995, GWRTAC is operated by Concurrent Technologies Corporation (CTC) in association with the University of Pittsburgh's Environmental Engineering Program through a Cooperative Agreement with the U.S. Environmental Protection Agency's (EPA) Technology Innovation Office (TIO). CTC, an independent nonprofit organization, is committed to assisting industry and government achieve world-class competitiveness. Through a unique concurrent engineering framework, CTC provides comprehensive solutions that improve product quality, productivity, and cost effectiveness.

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## TABLE OF CONTENTS

	<u>Page</u>
<b>1.0 SUMMARY</b>	<b>1</b>
<b>2.0 PROBLEM DESCRIPTION</b>	<b>2</b>
2.1 Metals at Contaminated Sites	2
2.2 Sources of Contaminants	3
2.2.1 Airborne Sources	3
2.2.2 Process Solid Wastes	3
2.2.3 Sludges	4
2.2.4 Soils	4
2.2.5 Direct Ground-Water Contamination	4
2.3 Definitions of Contaminant Concentrations	4
2.4 Chemical Fate and Mobility	5
2.4.1 Lead	7
2.4.2 Chromium	7
2.4.3 Arsenic	8
2.4.4 Zinc	9
2.4.5 Cadmium	9
2.4.6 Copper	10
2.4.7 Mercury	10
2.5 Influence of Soil Properties on Mobility	11
2.5.1 Chemical Properties	11
2.5.2 Physical Properties	13
<b>3.0 AVAILABLE TECHNOLOGIES AND PERFORMANCE</b>	<b>14</b>
3.1 Site Characterization and Establishment of Remediation Goals	14
3.2 General Remediation Approaches	14
3.2.1 Isolation	14
3.2.1.1 Capping	16
3.2.1.2 Subsurface Barriers	16
3.2.2 Immobilization	18
3.2.2.1 Solidification/Stabilization	18
3.2.2.2 Vitrification	20

---

## TABLE OF CONTENTS (cont.)

	<u>Page</u>
3.2.3 Toxicity and/or Mobility Reduction	21
3.2.3.1 Chemical Treatment	21
3.2.3.2 Permeable Treatment Walls	22
3.2.3.3 Biological Treatment	23
3.2.4 Physical Separation	25
3.2.5 Extraction	26
3.2.5.1 Soil Washing	26
3.2.5.2 Pyrometallurgical Extraction	28
3.2.5.3 <i>In Situ</i> Soil Flushing	29
3.2.5.4 Electrokinetic Treatment	29
3.3 Performance of Available Commercial Technologies	29
3.3.1 Superfund Innovative Technology Evaluation (SITE) Demonstration and Best Demonstrated Available Technology (BDAT) Status	30
3.3.2 Containment	30
3.3.3 Ex Situ Remediation	31
3.3.3.1 Solidification/Stabilization	32
3.3.3.2 Soil Washing	33
3.3.3.3 Vitrification	34
3.3.3.4 Pyrometallurgical Separation	35
3.3.4 <i>In Situ</i> Remediation	36
3.3.4.1 Solidification/Stabilization	36
3.3.4.2 Vitrification	37
3.3.4.3 <i>In Situ</i> Soil Flushing	37
3.3.4.4 Electrokinetic Extraction	39
3.3.4.5 Biological Treatment	40
3.4 Best Technology by Metal	40
<b>4.0 COST ESTIMATES</b>	<b>42</b>
4.1 Containment	42
4.2 Solidification/Stabilization	43
4.3 Vitrification	43

---

## TABLE OF CONTENTS (cont.)

	<u>Page</u>
4.4 Soil Washing	43
4.5 <i>In Situ</i> Soil Flushing	43
4.6 Electrokinetic Treatment	43
<b>5.0 REGULATORY/POLICY REQUIREMENTS AND ISSUES</b>	<b>44</b>
<b>6.0 LESSONS LEARNED AND TECHNOLOGY DIRECTIONS</b>	<b>46</b>
<b>7.0 REFERENCES</b>	<b>47</b>

---

## LIST OF FIGURES

<u>Figure No.</u>	<u>Title</u>	<u>Page</u>
1	Metals Most Commonly Present in all Matrices at Superfund Sites	2
2	Metal Adsorption to Hydrous Iron Oxide Gels	6
3	Typical pH edges for (a) cation sorption and (b) anion sorption	12
4	Estimated Operating Costs of Available Remediation Technologies for Metal-Contaminated Soils	42

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## LIST OF TABLES

<u>Table No.</u>	<u>Title</u>	<u>Page</u>
1	Remediation Technologies Matrix for Metals in Soil and Ground-Water	15
2	Example Containment Applications at Metals-Contaminated Superfund Sites	31
3	Example Solidification/Stabilization Applications at Selected Metals-Contaminated Superfund Sites	33
4	Example Soil Washing Applications at Metals-Contaminated Superfund Sites	34
5	Approximate Vapor Pressures and Glass Solubility Limits for Metals	35
6	Example <i>In Situ</i> Solidification/Stabilization Applications at Metals-Contaminated Superfund Sites	36
7	Example <i>In Situ</i> Vitrification Applications at Metals-Contaminated Superfund Sites	37
8	Example <i>In Situ</i> Soil Flushing Applications at Metals-Contaminated Superfund Sites	38
9	Example Electrokinetic Applications at Metals-Contaminated Sites	39
10	Summary of Best Demonstrated Available Technologies (BDATs) for RCRA Wastes	41
11	Examples of U.S. Cleanup Goals/Standards for Selected Metals in Soils	45
12	Examples of U.S. Cleanup Goals/Standards for Selected Metals in Groundwater	45

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## 1.0 SUMMARY

Metals contamination is a persistent problem at many contaminated sites. In the U.S., the most commonly occurring metals at Superfund sites are lead, chromium, arsenic, zinc, cadmium, copper, and mercury. The presence of metals in groundwater and soils can pose a significant threat to human health and ecological systems. The chemical form of the metal contaminant influences its solubility, mobility, and toxicity in ground-water systems. The chemical form of metals depends on the source of the metal waste and the soil and ground-water chemistry at the site. A detailed site characterization must be performed to assess the type and level of metals present and allow evaluation of remedial alternatives.

Typically metals are relatively immobile in subsurface systems as a result of precipitation or adsorption reactions. For this reason, remediation activities at metals-contaminated sites have focused on the solid-phase sources of metals, i.e., contaminated soils, sludges, wastes, or debris.

A range of technologies is available for remediation of metals-contaminated soil and groundwater at Superfund sites. General approaches to remediation of metal contamination include isolation, immobilization, toxicity reduction, physical separation and extraction. These general approaches can be used for many types of contaminants but the specific technology selected for treatment of a metals-contaminated site will depend on the form of the contamination and other site-specific characteristics. One or more of these approaches are often combined for more cost-effective treatment. A number of the available technologies have been demonstrated in full-scale applications and are presently commercially available. A comprehensive list of these technologies is available (U.S. EPA, 1996a). Several other technologies are being tested for application to metals-contaminated sites. This report summarizes remediation technologies for metals-contaminated soil and groundwater whose performance at full-scale has been verified under the United States Environmental Protection Agency (U.S. EPA) Superfund Innovative Technology Evaluation (SITE) program for evaluation of emerging and demonstrated technologies. The focus of this program is the demonstration phase in which the technologies are field-tested and performance and cost data are collected. Technologies available for treatment of metals-contaminated soil and groundwater by each of the general approaches to remediation are presented, and the applicability of these technologies to different types of metal contamination and physical site characteristics are evaluated. Cost ranges are provided for a number of the technologies. The most promising emerging technologies are also examined.

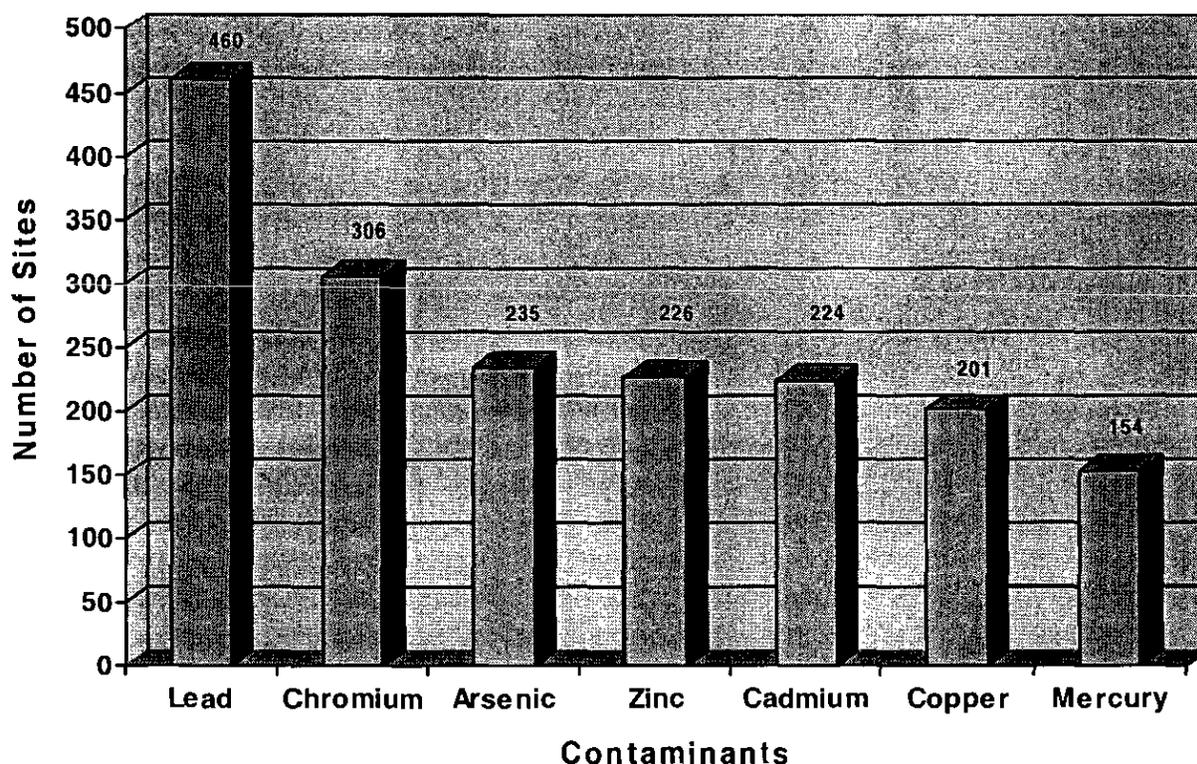
Treatment of metals-contaminated groundwater has typically involved flushing and above-ground treatment, while treatment of contaminated solids most often has been performed by excavation followed by ex situ treatment or disposal. The most common ex situ treatment for excavated soils is solidification/stabilization. *In situ* treatment methods for metals-contaminated soil and groundwater are being tested and will be applied with increasing frequency.

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## 2.0 PROBLEM DESCRIPTION

### 2.1 METALS AT CONTAMINATED SITES

Approximately 75% of Superfund sites for which Records of Decision (RODs) have been signed contain metals as a form of contamination. Some of these sites contain mixed metal-organic wastes for which metals might not be the primary contaminant of concern. The most common metals found at contaminated sites are (U.S. EPA, 1996b), in order: lead (Pb), chromium (Cr), arsenic (As), zinc (Zn), cadmium (Cd), copper (Cu), and mercury (Hg). Figure 1 summarizes the frequency with which these metals occur at Superfund sites.



**Figure 1.** Metals Most Commonly Present in all Matrices at Superfund Sites (from U.S. EPA, 1996)

The specific type of metal contamination found at a Superfund site is directly related to the operation that occurred at the site. The range of contaminant concentrations and the physical and chemical forms of contaminants will also depend on activities and disposal patterns for contaminated wastes on the site. Other factors that may influence the form, concentration and distribution of metal contaminants include soil and ground-water chemistry and local transport mechanisms.

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## **2.2 SOURCES OF CONTAMINANTS**

Surface water and groundwater may be contaminated with metals from wastewater discharges or by direct contact with metals-contaminated soils, sludges, mining wastes, and debris. Metal-bearing solids at contaminated sites can originate from a wide variety of sources in the form of airborne emissions, process solid wastes, sludges or spills. The contaminant sources influence the heterogeneity of contaminated sites on a macroscopic and microscopic scale. Variations in contaminant concentration and matrix influence the risks associated with metal contamination and treatment options.

### **2.2.1 Airborne Sources**

Airborne sources of metals include stack or duct emissions of air, gas, or vapor streams, and fugitive emissions such as dust from storage areas or waste piles. Metals from airborne sources are generally released as particulates contained in the gas stream. Some metals such as arsenic, cadmium, and lead can also volatilize during high-temperature processing. These metals will convert to oxides and condense as fine particulates unless a reducing atmosphere is maintained. (Smith et al., 1995)

Stack emissions can be distributed over a wide area by natural air currents until dry and/or wet precipitation mechanisms remove them from the gas stream. Fugitive emissions are often distributed over a much smaller area because emissions are made near the ground. In general, contaminant concentrations are lower in fugitive emissions compared to stack emissions. The type and concentration of metals emitted from both types of sources will depend on site-specific conditions.

### **2.2.2 Process Solid Wastes**

Process solid wastes can result from a variety of industrial processes. These metal-bearing solid wastes are disposed above ground in waste piles or below ground or under cover in landfills. Examples of process solid wastes include slags, fly ash, mold sands, abrasive wastes, ion exchange resins, spent catalysts, spent activated carbon, and refractory bricks (Zimmerman and Coles, 1992). The composition of the process waste influences the density, porosity, and leach resistance of the waste and must be considered in evaluating the contaminated matrix.

Because waste piles are above ground, they are exposed to weathering which can disperse the waste pile to the surrounding soil, water and air and can result in generation of leachate which infiltrates into the subsurface environment. The ability of landfills to contain process solid wastes varies due to the range of available landfill designs. Uncontained landfills can release contaminants into infiltrating surface water or groundwater or via wind and surface erosion.

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### **2.2.3 Sludges**

The composition of sludges depends on the original waste stream and the process from which it was derived. Sludges resulting from a uniform wastestream, such as wastewater treatment sludges, are typically more homogeneous and have more uniform matrix characteristics. Sludge pits, on the other hand, often contain a mixture of wastes that have been aged and weathered, causing a variety of reactions to occur. Sludge pits often require some form of pretreatment before wastes can be treated or recycled (Smith et al., 1995).

### **2.2.4 Soils**

Soil consists of a mixture of weathered minerals and varying amounts of organic matter. Soils can be contaminated as a result of spills or direct contact with contaminated waste streams such as airborne emissions, process solid wastes, sludges, or leachate from waste materials. The solubility of metals in soil is influenced by the chemistry of the soil and groundwater (Sposito, 1989; Evans, 1989). Factors such as pH, Eh, ion exchange capacity, and complexation/chelation with organic matter directly affect metal solubility.

### **2.2.5 Direct Ground-Water Contamination**

Groundwater can be contaminated with metals directly by infiltration of leachate from land disposal of solid wastes, liquid sewage or sewage sludge, leachate from mine tailings and other mining wastes, deep-well disposal of liquid wastes, seepage from industrial waste lagoons, or from other spills and leaks from industrial metal processing facilities (e.g., steel plants, plating shops, etc.). A variety of reactions may occur which influence the speciation and mobility of metal contaminants including acid/base, precipitation/dissolution, oxidation/reduction, sorption or ion exchange. Precipitation, sorption, and ion exchange reactions can retard the movement of metals in groundwater. The rate and extent of these reactions will depend on factors such as pH, Eh, complexation with other dissolved constituents, sorption and ion exchange capacity of the geological materials, and organic matter content. Ground-water flow characteristics also influence the transport of metal contaminants.

## **2.3 DEFINITIONS OF CONTAMINANT CONCENTRATIONS**

Sludges, soils, and solid wastes are multiphase materials which may contain metals in the solid, gaseous, or liquid phases. This complicates analysis and interpretation of reported results. For example, the most common method for determining the concentration of metals contaminants in soil is via total elemental analysis (U.S. EPA Method 3050). The level of metal contamination determined by this method is expressed as mg metal/kg soil. This analysis does not specify requirements for the moisture content of the soil and may therefore include soil water. This measurement may also be reported on a dry soil basis.

The level of contamination may also be reported as leachable metals as determined by leach tests, such as the toxicity characteristic leaching procedure, or TCLP test (U.S. EPA

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Method 1311) or the synthetic precipitation leaching procedure, or SPLP test (U.S. EPA Method 1312). These procedures measure the concentration of metals in leachate from soil contacted with an acetic acid solution (TCLP) or a dilute solution of sulfuric and nitric acid (SPLP). In this case, metal contamination is expressed in mg/L of the leachable metal.

Other types of leaching tests have been proposed (see summary by Environment Canada, 1990), including sequential extraction procedures (Tessier et al., 1979) and extraction of acid volatile sulfide (DiToro et al., 1992). Sequential procedures contact the solid with a series of extractant solutions that are designed to dissolve different fractions of the associated metal. These tests may provide insight into the different forms of metal contamination present (e.g., see Van Benschoten et al., 1994).

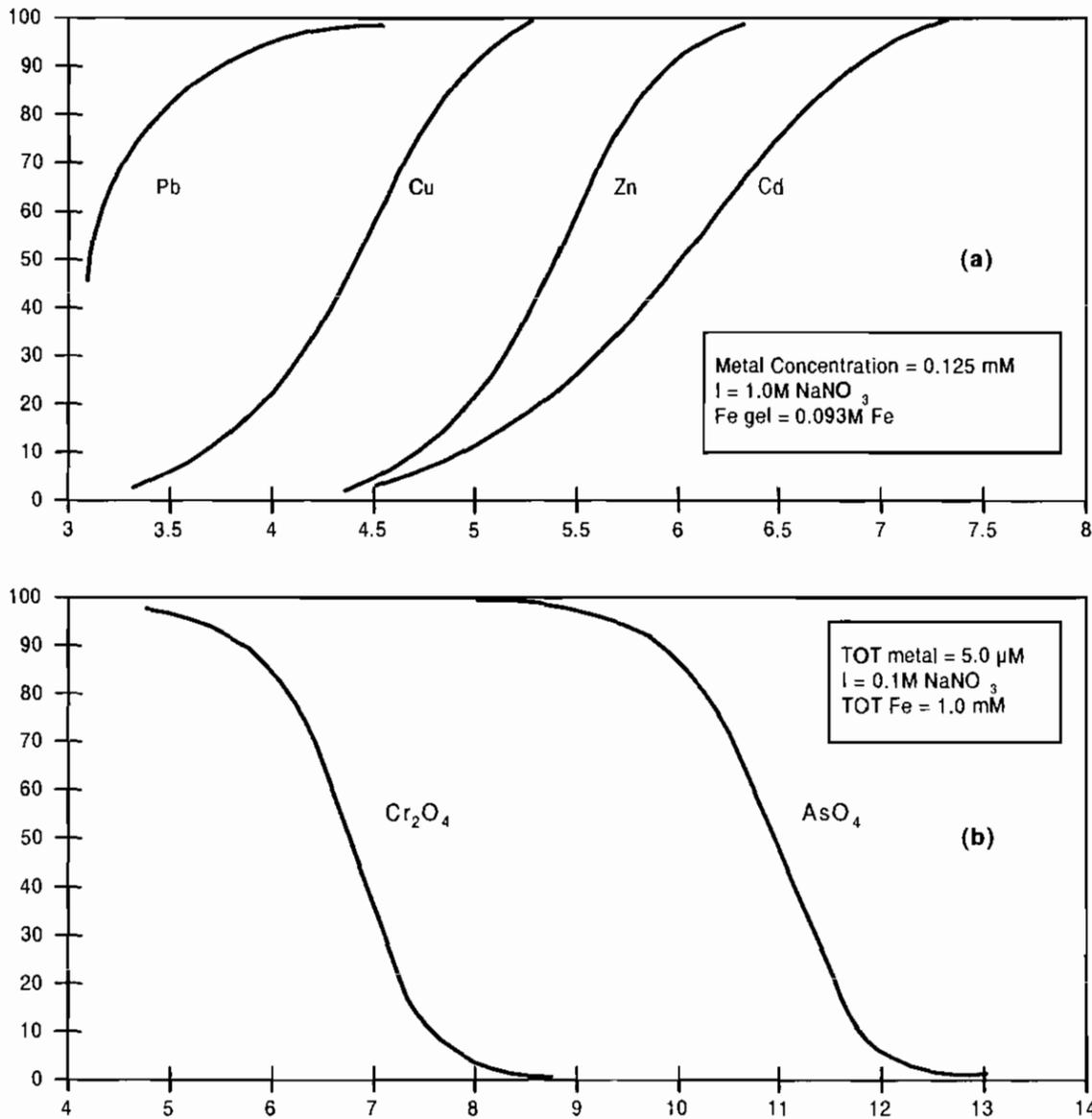
Contaminant concentrations can be measured directly in metals-contaminated water. These concentrations are most commonly expressed as total dissolved metals in mass concentrations (mg/L or  $\mu\text{g/L}$ ) or in molar concentrations (moles/L). In dilute solutions, a mg/L is equivalent to one part per million (ppm), and a  $\mu\text{g/L}$  is equivalent to one part per billion (ppb).

Ground-water samples are usually filtered with a 0.45  $\mu\text{m}$  filter prior to analysis for metals, though this is not always required and has recently been prohibited by many states and some U.S. EPA programs that require analysis of total metals. Interest in measurement of total metal concentrations (dissolved and particulate-associated metals) usually derives from concern about possible transport of metals adsorbed on mobile colloidal particles (e.g., Kaplan et al., 1995). Research indicates that significant colloid-facilitated transport of metals can occur only under a fairly specialized set of conditions (Roy and Dzombak, 1997), but the conservative approach in monitoring system design is to try to capture any mobile colloids present. The problem with sampling groundwater without filtration is that particles from the well material, well slime coatings, or well pack may be sampled, and any subsequent analysis will not accurately reflect ground-water composition. To avoid such artifacts, but still permit sampling that can capture any mobile colloids present in the groundwater, monitoring wells are purged before sampling to remove the casing water and obtain representative ground-water samples. Low-flow purging and sampling techniques have been developed to minimize sample disturbances that may affect analysis (Puls, 1994; Puls and Paul, 1995).

## **2.4 CHEMICAL FATE AND MOBILITY**

The fate and transport of a metal in soil and groundwater depends significantly on the chemical form and speciation of the metal (Allen et al., 1991). The mobility of metals in ground-water systems is hindered by reactions that cause metals to adsorb or precipitate, or chemistry that tends to keep metals associated with the solid phase and prevent them from dissolving. These mechanisms can retard the movement of metals and also provide a long-term source of metal contaminants (NRC, 1994). While the various metals undergo similar reactions in a number of aspects, the extent and nature of these reactions varies

under particular conditions. In Figure 2, for example, the extent of sorption of several metal cations and anions onto iron oxide is shown as a function of pH for a particular background electrolyte composition. It may be seen there that lead sorbs extensively at much lower pH values than zinc or cadmium (Kinniburgh et al., 1976).



**Figure 2.** Metal Adsorption to Hydrous Iron Oxide Gels (a) Metal Cations (adapted from Kinniburgh et al., 1976) and (b) Metal Anions (adapted from Leckie et al., 1980; Honeyman et al., 1984)

The chemical form and speciation of some of the more important metals found at contaminated sites are discussed below. The influence of chemical form on fate and mobility of these compounds is also discussed.

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## 2.4.1 Lead

The primary industrial sources of lead (Pb) contamination include metal smelting and processing, secondary metals production, lead battery manufacturing, pigment and chemical manufacturing, and lead-contaminated wastes. Widespread contamination due to the former use of lead in gasoline is also of concern. Lead released to groundwater, surface water and land is usually in the form of elemental lead, lead oxides and hydroxides, and lead-metal oxyanion complexes (Smith et al., 1995).

Lead occurs most commonly with an oxidation state of 0 or +II. Pb(II) is the more common and reactive form of lead and forms mononuclear and polynuclear oxides and hydroxides. Under most conditions  $Pb^{2+}$  and lead-hydroxy complexes are the most stable forms of lead (Smith et al., 1995). Low solubility compounds are formed by complexation with inorganic ( $Cl^-$ ,  $CO_3^{2-}$ ,  $SO_4^{2-}$ ,  $PO_4^{3-}$ ) and organic ligands (humic and fulvic acids, EDTA, amino acids) (Bodek et al., 1988). Lead carbonate solids form above pH 6 and  $PbS$  is the most stable solid when high sulfide concentrations are present under reducing conditions.

Most lead that is released to the environment is retained in the soil (Evans, 1989). The primary processes influencing the fate of lead in soil include adsorption, ion exchange, precipitation, and complexation with sorbed organic matter. These processes limit the amount of lead that can be transported into the surface water or groundwater. The relatively volatile organolead compound tetramethyl lead may form in anaerobic sediments as a result of alkylation by microorganisms (Smith et al., 1995).

The amount of dissolved lead in surface water and groundwater depends on pH and the concentration of dissolved salts and the types of mineral surfaces present. In surface water and ground-water systems, a significant fraction of lead is undissolved and occurs as precipitates ( $PbCO_3$ ,  $Pb_2O$ ,  $Pb(OH)_2$ ,  $PbSO_4$ ), sorbed ions or surface coatings on minerals, or as suspended organic matter.

## 2.4.2 Chromium

Chromium(Cr) is one of the less common elements and does not occur naturally in elemental form, but only in compounds. Chromium is mined as a primary ore product in the form of the mineral chromite,  $FeCr_2O_4$ . Major sources of Cr contamination include releases from electroplating processes and the disposal of chromium containing wastes (Smith et al., 1995).

Cr(VI) is the form of chromium commonly found at contaminated sites. Chromium can also occur in the +III oxidation state, depending on pH and redox conditions. Cr (VI) is the dominant form of chromium in shallow aquifers where aerobic conditions exist. Cr(VI) can be reduced to Cr(III) by soil organic matter,  $S^{2-}$  and  $Fe^{2+}$  ions under anaerobic conditions often encountered in deeper groundwater. Major Cr(VI) species include chromate ( $CrO_4^{2-}$ ) and dichromate ( $Cr_2O_7^{2-}$ ) which precipitate readily in the presence of metal cations

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(especially  $\text{Ba}^{2+}$ ,  $\text{Pb}^{2+}$ , and  $\text{Ag}^+$ ). Chromate and dichromate also adsorb on soil surfaces, especially iron and aluminum oxides. Cr(III) is the dominant form of chromium at low pH (<4).  $\text{Cr}^{3+}$  forms solution complexes with  $\text{NH}_3$ ,  $\text{OH}^-$ ,  $\text{Cl}^-$ ,  $\text{F}^-$ ,  $\text{CN}^-$ ,  $\text{SO}_4^{2-}$ , and soluble organic ligands. Cr(VI) is the more toxic form of chromium and is also more mobile. Cr(III) mobility is decreased by adsorption to clays and oxide minerals below pH 5 and low solubility above pH 5 due to the formation of  $\text{Cr}(\text{OH})_3(\text{s})$  (Chrotowski et al., 1991).

Chromium mobility depends on sorption characteristics of the soil, including clay content, iron oxide content and the amount of organic matter present. Chromium can be transported by surface runoff to surface waters in its soluble or precipitated form. Soluble and unadsorbed chromium complexes can leach from soil into groundwater. The leachability of Cr(VI) increases as soil pH increases. Most of chromium released into natural waters is particle associated, however, and is ultimately deposited into the sediment (Smith et al., 1995).

### 2.4.3 Arsenic

Arsenic (As) is a semimetallic element that occurs in a wide variety of minerals, mainly as  $\text{As}_2\text{O}_3$ , and can be recovered from processing of ores containing mostly copper, lead, zinc, silver and gold. It is also present in ashes from coal combustion. Arsenic exhibits fairly complex chemistry and can be present in several oxidation states (-III, 0, III, V) (Smith et al., 1995).

In aerobic environments, As(V) is dominant, usually in the form of arsenate ( $\text{AsO}_4^{3-}$ ) in various protonation states:  $\text{H}_3\text{AsO}_4$ ,  $\text{H}_2\text{AsO}_4^-$ ,  $\text{HAsO}_4^{2-}$ ,  $\text{AsO}_4^{3-}$ . Arsenate, and other anionic forms of arsenic behave as chelates and can precipitate when metal cations are present (Bodek et al., 1988). Metal arsenate complexes are stable only under certain conditions. As(V) can also coprecipitate with or adsorb onto iron oxyhydroxides under acidic and moderately reducing conditions. Coprecipitates are immobile under these conditions but arsenic mobility increases as pH increases (Smith et al., 1995).

Under reducing conditions As(III) dominates, existing as arsenite ( $\text{AsO}_3^{3-}$ ) and its protonated forms:  $\text{H}_3\text{AsO}_3$ ,  $\text{H}_2\text{AsO}_3^-$ ,  $\text{HAsO}_3^{2-}$ . Arsenite can adsorb or coprecipitate with metal sulfides and has a high affinity for other sulfur compounds. Elemental arsenic and arsine,  $\text{AsH}_3$ , may be present under extreme reducing conditions. Biotransformation (via methylation) of arsenic creates methylated derivatives of arsine, such as dimethyl arsine  $\text{HAs}(\text{CH}_3)_2$  and trimethylarsine  $\text{As}(\text{CH}_3)_3$  which are highly volatile.

Since arsenic is often present in anionic form, it does not form complexes with simple anions such as  $\text{Cl}^-$  and  $\text{SO}_4^{2-}$ . Arsenic speciation also includes organometallic forms such as methylarsinic acid  $(\text{CH}_3)\text{AsO}_2\text{H}_2$  and dimethylarsinic acid  $(\text{CH}_3)_2\text{AsO}_2\text{H}$ .

Many arsenic compounds sorb strongly to soils and are therefore transported only over short distances in groundwater and surface water. Sorption and coprecipitation with hydrous

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iron oxides are the most important removal mechanisms under most environmental conditions (Krause and Ettel, 1989; Pierce and Moore, 1982). Arsenates can be leached easily if the amount of reactive metal in the soil is low. As(V) can also be mobilized under reducing conditions that encourage the formation of As(III), under alkaline and saline conditions, in the presence of other ions that compete for sorption sites, and in the presence of organic compounds that form complexes with arsenic (Smith et al., 1995).

#### 2.4.4 Zinc

Zinc (Zn) does not occur naturally in elemental form. It is usually extracted from mineral ores to form zinc oxide (ZnO). The primary industrial use for Zinc is as a corrosion-resistant coating for iron or steel (Smith et al., 1995).

Zinc usually occurs in the +II oxidation state and forms complexes with a number of anions, amino acids and organic acids. Zn may precipitate as  $Zn(OH)_2(s)$ ,  $ZnCO_3(s)$ ,  $ZnS(s)$ , or  $Zn(CN)_2(s)$ .

Zinc is one of the most mobile heavy metals in surface waters and groundwater because it is present as soluble compounds at neutral and acidic pH values. At higher pH values, zinc can form carbonate and hydroxide complexes which control zinc solubility. Zinc readily precipitates under reducing conditions and in highly polluted systems when it is present at very high concentrations, and may coprecipitate with hydrous oxides of iron or manganese (Smith et al., 1995).

Sorption to sediments or suspended solids, including hydrous iron and manganese oxides, clay minerals, and organic matter, is the primary fate of zinc in aquatic environments. Sorption of zinc increases as pH increases and salinity decreases.

#### 2.4.5 Cadmium

Cadmium (Cd) occurs naturally in the form of CdS or  $CdCO_3$ . Cadmium is recovered as a by-product from the mining of sulfide ores of lead, zinc and copper. Sources of cadmium contamination include plating operations and the disposal of cadmium-containing wastes (Smith et al., 1995).

The form of cadmium encountered depends on solution and soil chemistry as well as treatment of the waste prior to disposal. The most common forms of cadmium include  $Cd^{2+}$ , cadmium-cyanide complexes, or  $Cd(OH)_2$  solid sludge (Smith et al., 1995). Hydroxide ( $Cd(OH)_2$ ) and carbonate ( $CdCO_3$ ) solids dominate at high pH whereas  $Cd^{2+}$  and aqueous sulfate species are the dominant forms of cadmium at lower pH (<8). Under reducing conditions when sulfur is present, the stable solid CdS(s) is formed. Cadmium will also precipitate in the presence of phosphate, arsenate, chromate and other anions, although solubility will vary with pH and other chemical factors.

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Cadmium is relatively mobile in surface water and ground-water systems and exists primarily as hydrated ions or as complexes with humic acids and other organic ligands (Callahan et al., 1979). Under acidic conditions, cadmium may also form complexes with chloride and sulfate. Cadmium is removed from natural waters by precipitation and sorption to mineral surfaces, especially oxide minerals, at higher pH values (>pH 6). Removal by these mechanisms increases as pH increases. Sorption is also influenced by the cation exchange capacity (CEC) of clays, carbonate minerals, and organic matter present in soils and sediments. Under reducing conditions, precipitation as CdS controls the mobility of cadmium (Smith et al., 1995).

#### 2.4.6 Copper

Copper (Cu) is mined as a primary ore product from copper sulfide and oxide ores. Mining activities are the major source of copper contamination in groundwater and surface waters. Other sources of copper include algicides, chromated copper arsenate (CCA) pressure-treated lumber, and copper pipes.

Solution and soil chemistry strongly influence the speciation of copper in ground-water systems. In aerobic, sufficiently alkaline systems,  $\text{CuCO}_3$  is the dominant soluble copper species. The cupric ion,  $\text{Cu}^{2+}$ , and hydroxide complexes,  $\text{CuOH}^+$  and  $\text{Cu}(\text{OH})_2$ , are also commonly present. Copper forms strong solution complexes with humic acids. The affinity of Cu for humates increases as pH increases and ionic strength decreases. In anaerobic environments, when sulfur is present  $\text{CuS}(\text{s})$  will form.

Copper mobility is decreased by sorption to mineral surfaces.  $\text{Cu}^{2+}$  sorbs strongly to mineral surfaces over a wide range of pH values (Dzombak and Morel, 1990).

The cupric ion ( $\text{Cu}^{2+}$ ) is the most toxic species of copper. Copper toxicity has also been demonstrated for  $\text{CuOH}^+$  and  $\text{Cu}_2(\text{OH})_2^{2+}$  (LaGrega et al., 1994).

#### 2.4.7 Mercury

The primary source of mercury is the sulfide ore cinnabar. Mercury (Hg) is usually recovered as a by-product of ore processing (Smith et al., 1995). Release of mercury from coal combustion is a major source of mercury contamination. Releases from manometers at pressure measuring stations along gas/oil pipelines also contribute to mercury contamination.

After release to the environment, mercury usually exists in mercuric ( $\text{Hg}^{2+}$ ), mercurous ( $\text{Hg}_2^{2+}$ ), elemental ( $\text{Hg}^0$ ), or alkylated form (methyl/ethyl mercury). The redox potential and pH of the system determine the stable forms of mercury that will be present. Mercurous and mercuric mercury are more stable under oxidizing conditions. When mildly reducing conditions exist, organic or inorganic mercury may be reduced to elemental mercury, which may then be converted to alkylated forms by biotic or abiotic processes. Mercury is most

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toxic in its alkylated forms which are soluble in water and volatile in air (Smith et al., 1995).

Hg(II) forms strong complexes with a variety of both inorganic and organic ligands, making it very soluble in oxidized aquatic systems (Bodek et al., 1988). Sorption to soils, sediments, and humic materials is an important mechanism for removal of mercury from solution. Sorption is pH-dependent and increases as pH increases. Mercury may also be removed from solution by coprecipitation with sulfides (Smith et al., 1995).

Under anaerobic conditions, both organic and inorganic forms of mercury may be converted to alkylated forms by microbial activity, such as by sulfur-reducing bacteria. Elemental mercury may also be formed under anaerobic conditions by demethylation of methyl mercury, or by reduction of Hg(II). Acidic conditions (pH<4) also favor the formation of methyl mercury, whereas higher pH values favor precipitation of HgS(s) (Smith et al., 1995).

## **2.5 INFLUENCE OF SOIL PROPERTIES ON MOBILITY**

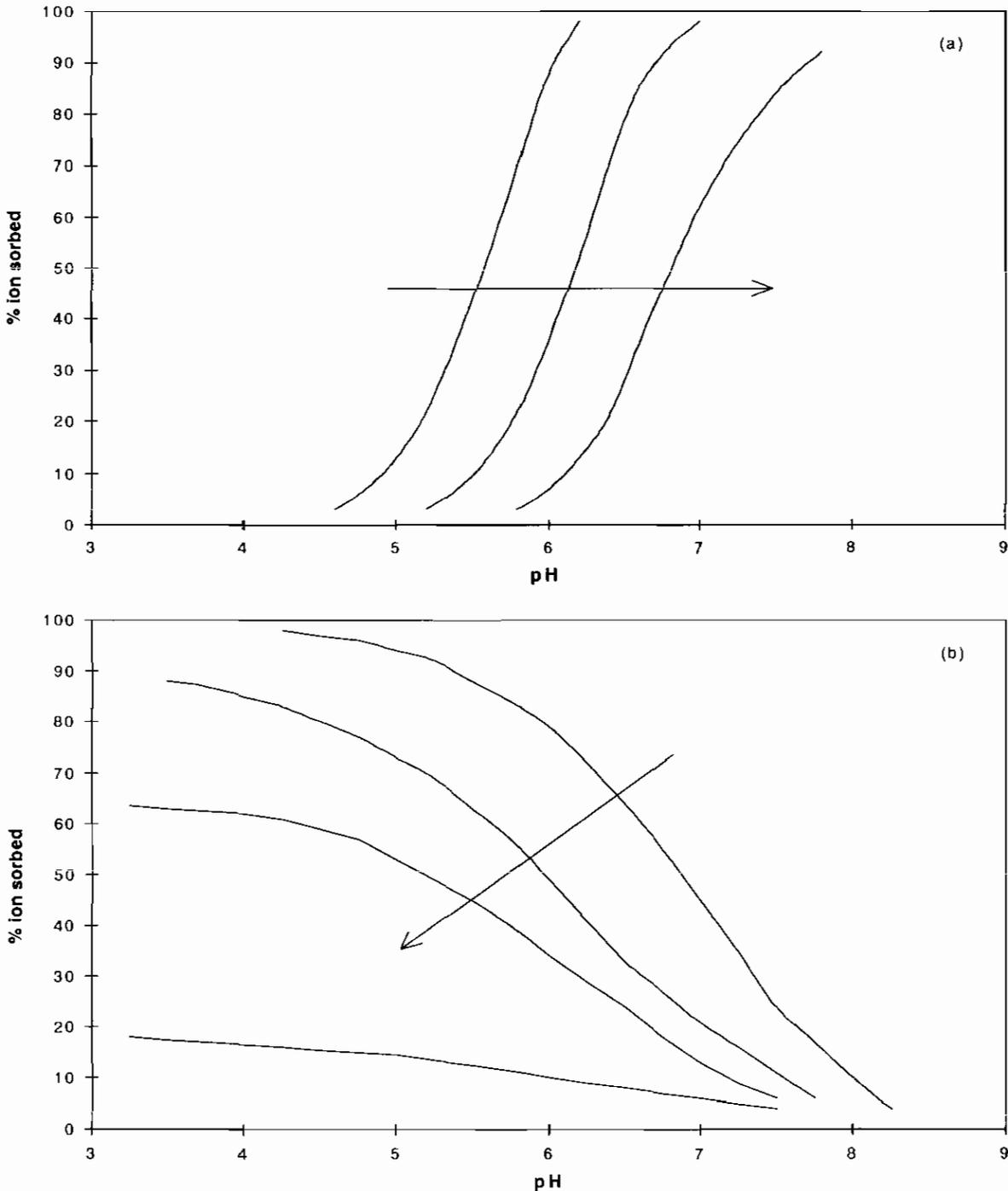
Chemical and physical properties of the contaminated matrix influence the mobility of metals in soils and groundwater. Contamination exists in three forms in the soil matrix: solubilized contaminants in the soil moisture, adsorbed contaminants on soil surfaces, and contaminants fixed chemically as solid compounds. The chemical and physical properties of the soil will influence the form of the metal contaminant, its mobility, and the technology selected for remediation (Gerber et al., 1991).

### **2.5.1 Chemical Properties**

The presence of inorganic anions (carbonate, phosphate, sulfide) in the soil water can influence the soil's ability to fix metals chemically. These anions can form relatively insoluble complexes with metal ions and cause metals to desorb and/or precipitate in their presence.

Soil pH values generally range between 4.0 and 8.5 with buffering by Al at low pH and by CaCO<sub>3</sub> at high pH (Wild, 1988). Metal cations are most mobile under acidic conditions while anions tend to sorb to oxide minerals in this pH range (Dzombak and Morel, 1987). At high pH, cations precipitate or adsorb to mineral surfaces and metal anions are mobilized. The presence of hydrous metal oxides of Fe, Al, Mn can strongly influence metal concentrations because these minerals can remove cations and anions from solution by ion exchange, specific adsorption and surface precipitation (Ellis and Fogg, 1985; Dzombak and Morel, 1987). As noted in the previous section, sorption of metal cations onto hydrous oxides generally increases sharply with pH and is most significant at pH values above the neutral range, while sorption of metal anions is greatest at low pH and decreases as pH is increased (Figure 3). Cation exchange capacity (CEC) refers to the concentration of readily exchangeable cations on a mineral surface and is often used to indicate the affinity of soils for uptake of cations such as metals. Anion exchange capacity (AEC) indicates the affinity of soils for uptake of anions, and is usually significantly lower than the CEC of the soil. In addition to hydrous oxides, clays are also important ion exchange materials for metals

(Sposito, 1989). The presence of natural organic matter (NOM) has been shown to influence the sorption of metal ions to mineral surfaces. NOM has been observed to enhance sorption of  $\text{Cu}^{2+}$  at low pH, and suppress  $\text{Cu}^{2+}$  sorption at high pH (Tipping et al., 1983; Davis, 1984).



**Figure 3.** Typical pH edges for (a) cation sorption and (b) anion sorption. Arrows indicate direction of increasing sorbate/sorbent ratio. (From Dzombak, and Morel, 1990)

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Organic matter, particularly humic materials, can complex metals and affect their removal from solution (Ali and Dzombak, 1996). Humic materials contain carboxylic and phenolic functional groups that can complex with metal ions.

## 2.5.2 Physical Properties

Particle size distribution can influence the level of metal contamination in a soil. Fine particles (<100  $\mu\text{m}$ ) are more reactive and have a higher surface area than coarser material. As a result, the fine fraction of a soil often contains the majority of contamination. The distribution of particle sizes with which a metal contaminant is associated can determine the effectiveness of a number of metal remediation technologies, e.g., soil washing (Dzombak et al., 1994).

Soil moisture influences the chemistry of contaminated soil. The amount of dissolved minerals, pH and redox potential of the soil water depends on the soil moisture content.

Soil structure describes the size, shape, arrangement and degree of development of soils into structural units. Soil structure can influence contaminant mobility by limiting the degree of contact between groundwater and contaminants.

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## **3.0 AVAILABLE TECHNOLOGIES AND PERFORMANCE**

### **3.1 SITE CHARACTERIZATION AND ESTABLISHMENT OF REMEDIATION GOALS**

The physical and chemical form of the metal contaminant in soil or water strongly influences the selection of the appropriate remediation treatment approach. Information about the physical characteristics of the site and the type and level of contamination at the site must be obtained to enable accurate assessment of site contamination and remedial alternatives. The importance of adequate, well-planned site characterization to selection of an appropriate cost-effective remediation approach has been discussed many times (e.g., CII, 1995) but cannot be overemphasized. The contamination in the groundwater and soil should be characterized to establish the type, amount, and distribution of contaminants across different media.

Once the site has been characterized, the desired level of each contaminant in soil and groundwater must be determined. This is done by comparison of observed contaminant concentrations with soil and ground-water quality standards for a particular regulatory domain, or by performance of a site-specific risk assessment. Remediation goals for metals may be set as desired concentrations in groundwater, as total metal concentration in soil, as leachable metal in soil, or as some combination of these.

### **3.2 GENERAL REMEDIATION APPROACHES**

Several technologies exist for the remediation of metals-contaminated soil and water. These technologies are contained within five categories of general approaches to remediation: isolation, immobilization, toxicity reduction, physical separation and extraction. These are the same general approaches used for many types of contaminants in the subsurface (LaGrega et al., 1994). As is usually the case, combinations of one or more of these approaches are often used for more cost-effective treatment of a contaminated site. Table 1 summarizes key factors discussed in this report that were found to influence the applicability and selection of available remediation technologies.

#### **3.2.1 Isolation**

Isolation technologies attempt to prevent the transport of contaminants by containing them within a designated area. These technologies can be used to prevent further contamination of groundwater when other treatment options are not physically or economically feasible for a site. Contaminated sites may also be isolated temporarily in order to limit transport during site assessment and site remediation.

**Table 1. Remediation Technologies Matrix for Metals in Soils and Ground-Water**

Remediation Technology	Metals Treated	Cost	Long-term Effectiveness/ Permanence	Commercial Availability	General Acceptance	Applicability to High Metals Concentrations	Applicability to Mixed Waste (metals & organics)	Toxicity Reduction	Mobility Reduction	Volume Reduction
Capping	1-3	+	«	+	+	«	+	«	+	«
Subsurface Barriers	1-3,5	+	«	+	+	«	+	«	+	«
Solidification/ Stabilization Ex situ	1-3,5	•	•	+	+	+	+	«	+	«
Solidification/ Stabilization In situ	1,2,4,6	+	•	+	+	+	+	«	+	«
Vitrification Ex situ	1-3,5	«	+	•	•	+	+	«	+	«
Vitrification In situ	1-3,7	«	+	•	•	+	+	«	+	«
Chemical Treatment	2	-	•	•	•	-	-	+	+	«
Permeable Treatment Walls	2	-	•	•	•	-	-	+	+	«
Biological Treatment	1-5	+	«	•	•	«	-	+	+	«
Physical Separation	1-6	•	+	+	+	+	«	«	«	+
Soil Washing	1-3,5-7	•	+	+	+	+	•	«	«	+
Pyrometallurgical Extraction	1-5,7	«	+	+	+	+	«	«	«	+
In situ Soil Flushing	1,2,7	+	«	+	+	+	+	«	«	+
Electrokinetic Treatment	1-6	•	+	+	+	+	-	«	«	+

1-Lead, 2-Chromium, 3-Arsenic, 4-Zinc, 5-Cadmium, 6-Copper, 7-Mercury

+ Good, • Average, « Marginal, - Inadequate Information

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### **3.2.1.1 Capping**

Capping systems are used to provide an impermeable barrier to surface water infiltration to contaminated soil for prevention of further release of contaminants to the surrounding surface water or groundwater. Secondary objectives include controlling gas and odor emissions, improving aesthetics, and providing a stable surface over a contaminated site. Capping also eliminates risks associated with dermal contact and/or incidental ingestion of surface soils, but if this is the primary goal for the site and surface water infiltration is not a concern, a less expensive permeable cover may be preferred.

Capping provides a range of design options that includes simple single-layer caps and more complex multilayer systems (Rumer and Ryan, 1995; U.S. EPA, 1991). Design selection depends on site characteristics, remedial objectives and risk factors associated with the site. A variety of materials are available for use in capping systems and choice of materials is site specific because local soils are often incorporated into parts of the cap. Synthetic membranes such as high-density polyethylene are also available for incorporation into capping systems. Surface water controls, such as ditches and dikes are usually included to help control drainage from the cap. Multilayered capping systems may also include a hard cover and/or a layer of topsoil to separate the underlying layers from the ground surface. Revegetation is promoted in order to reinforce the topsoil, to reduce soil erosion and runoff velocity, and to help remove water from the soil by evapotranspiration (Rumer and Ryan, 1995).

### **3.2.1.2 Subsurface Barriers**

Subsurface barriers may be used to isolate contaminated soil and water by controlling the movement of groundwater at a contaminated site. These barriers are designed to reduce the movement of contaminated groundwater from the site, or to restrict the flow of uncontaminated groundwater through the contaminated site (Rumer and Ryan, 1995).

Vertical barriers are commonly used to restrict the lateral flow of groundwater. For effective isolation of the contaminated matrix, the barrier should extend and key into a continuous, low-permeability layer, such as clay or competent bedrock, below the contaminated area (U.S. EPA, 1985; Rumer and Ryan, 1995). If an impermeable layer is not available, a ground-water extraction system must be used to prevent transport of contaminants under the barrier. Vertical barriers may be installed upstream, downstream, or completely surrounding the site and are often implemented in conjunction with a capping system to control surface water infiltration. The use of circumferential barriers can prevent the escape of contamination from the site by using an infiltration barrier and collection system to create a hydraulic gradient in the inward direction. Vertical barriers are often limited to depths achievable with backhoe excavation technology for trenches, i.e., to about 30 feet (U.S. EPA, 1985).

Slurry walls are usually constructed in a vertical trench excavated under a slurry that is designed to prevent collapse and to form a filter cake on the walls of the trench to prevent

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the loss of fluids to the surrounding soil (Xanthakos, 1979). A vibrating beam method (Slurry Systems, Inc.) is also available in which the beam penetrates the ground and slurry materials are injected into the soil (with assistance from a high pressure/low volume jet if needed). Two options exist for the slurry composition. The soil-bentonite (SB) slurry wall is the most common type, and comprises a bentonite-water slurry that is mixed with a soil engineered to harden upon addition to the slurry (Rumer and Ryan, 1995). The trench can also be excavated under a portland cement-bentonite-water slurry that is left to harden and form a cement-bentonite (CB) slurry wall (LaGrega et al., 1994). Available technologies for installation of slurry walls allow installation to depths up to 125 feet.

Slurry walls are the most common type of vertical barrier due to their low relative cost. The use of slurry walls can be limited by the topography, geology, and type of contamination at the site. For example, an SB slurry will flow unless the site and confining layer are nearly level. Also, some contaminants, such as concentrated organics and strong acids/bases, can degrade SB materials and prevent the application of SB slurry walls at some sites (Rumer and Ryan, 1995).

Other available vertical barriers include grout curtains and sheet piles. Grout curtains are constructed by drilling a borehole and injecting a fluid into the surrounding soil that is designed to solidify and reduce water flow through the contaminated region (U.S. EPA, 1985). The fluid is pressure-injected in rows of staggered boreholes that are designed to overlap once the fluid has permeated into the surrounding soil. Common materials used to construct grout curtains include cement, clays, alkali-silicate, and organic polymers (Rumer and Ryan, 1995). Clays are the most widely used grouting materials due to their low cost. This technique is more expensive than slurry walls and its use is therefore usually limited to sealing voids in existing rock.

Sheet piles usually comprise steel pilings that are driven into the formation to create a wall to contain the groundwater. Sheet piles are seldom used at contaminated sites due to concerns about wall integrity. This method is generally limited to isolation of shallow contamination (40-50 ft) distributed over a relatively small area (U.S. EPA, 1985), or used in conjunction with a soil-bentonite slurry when site conditions prevent the use of conventional slurry walls (Rumer and Ryan, 1995).

Technologies for the construction of horizontal barriers are under investigation. Horizontal barriers would enable control of the downward migration of contaminants by lining the site without requiring excavation of the contaminated matrix. The technologies under investigation include grout injection by vertical boring and horizontal drilling. The vertical boring method is similar to the construction of grout curtains except that the grout is injected at a fixed elevation over a tightly spaced grid of vertical boreholes to create an impermeable horizontal layer. Problems with this method include soil compaction by the large drill rigs situated over the contaminated area. Also, the vertical boreholes would provide access to the deeper layers and may therefore increase vertical migration of contaminants. Horizontal drilling involves the use of directional drilling techniques to create the horizontal grout layer.

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Horizontal barriers may also be used in conjunction with vertical barriers at sites where a natural aquitard is not present. In this case, the vertical barrier could key into the horizontal barrier to prevent the transport of contaminants under the vertical barrier (Smith et al., 1995).

### **3.2.2 Immobilization**

Immobilization technologies are designed to reduce the mobility of contaminants by changing the physical or leaching characteristics of the contaminated matrix. Mobility is usually decreased by physically restricting contact between the contaminant and the surrounding groundwater, or by chemically altering the contaminant to make it more stable with respect to dissolution in groundwater. The aqueous and solid phase chemistry of metals is conducive to immobilization by these techniques. A variety of methods are available for immobilization of metal contaminants, including those that use chemical reagents and/or thermal treatment to physically bind the contaminated soil or sludge. Most immobilization technologies can be performed *ex situ* or *in situ*. *In situ* processes are preferred due to the lower labor and energy requirements, but implementation *in situ* will depend on specific site conditions.

#### **3.2.2.1 Solidification/Stabilization**

Solidification and stabilization (S/S) immobilization technologies are the most commonly selected treatment options for metals-contaminated sites (Conner, 1990). Solidification involves the formation of a solidified matrix that physically binds the contaminated material. Stabilization, also referred to as fixation, usually utilizes a chemical reaction to convert the waste to a less mobile form. The general approach for solidification/stabilization treatment processes involves mixing or injecting treatment agents to the contaminated soils. Inorganic binders, such as cement, fly ash, or blast furnace slag, and organic binders such as bitumen are used to form a crystalline, glassy or polymeric framework around the waste. The dominant mechanism by which metals are immobilized is by precipitation of hydroxides within the solid matrix (Bishop et al., 1982; Shively et al., 1986).

S/S technologies are not useful for some forms of metal contamination, such as species that exist as anions (e.g., Cr(VI), arsenic) or metals that don't have low-solubility hydroxides (e.g., mercury). S/S may not be applicable at sites containing wastes that include organic forms of contamination, especially if volatile organics are present. Mixing and heating associated with binder hydration may release organic vapors. Pretreatment, such as air stripping or incineration, may be used to remove the organics and prepare the waste for metal stabilization/solidification (Smith et al., 1995). The application of S/S technologies will also be affected by the chemical composition of the contaminated matrix, the amount of water present, and the ambient temperature. These factors can interfere with the solidification/stabilization process by inhibiting bonding of the waste to the binding material, retarding the setting of the mixtures, decreasing the stability of the matrix, or reducing the strength of the solidified area (U.S. EPA, 1990b).

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Cement-based binders and stabilizers are common materials used for implementation of S/S technologies (Conner, 1990). Portland cement, a mixture of Ca-silicates, aluminates, aluminoferrites, and sulfates is an important cement-based material. Pozzolanic materials which consist of small spherical particles formed by coal combustion (such as fly ash) and in lime and cement kilns, are also commonly used for S/S. Pozzolans exhibit cement-like properties, especially if the silica content is high. Portland cement and pozzolans can be used alone or together to obtain optimal properties for a particular site (U.S. EPA, 1989).

Organic binders may also be used to treat metals through polymer microencapsulation. This process uses organic materials such as bitumen, polyethylene, paraffins, waxes and other polyolefins as thermoplastic or thermosetting resins. For polymer encapsulation, the organic materials are heated and mixed with the contaminated matrix at elevated temperatures (120° to 200°C). The organic materials polymerize, agglomerate the waste and the waste matrix is encapsulated (U.S. EPA, 1989). Organics are volatilized and collected and the treated material is extruded for disposal or possible reuse (e.g., as paving material) (Smith et al., 1995). The contaminated material may require pretreatment to separate rocks and debris and dry the feed material. Polymer encapsulation requires more energy and more complex equipment than cement-based S/S operations. Bitumen (asphalt) is the cheapest and most common thermoplastic binder (U.S. EPA, 1989).

S/S is achieved by mixing the contaminated material with appropriate amounts of binder/stabilizer and water. The mixture sets and cures to form a solidified matrix and contain the waste. The cure time and pour characteristics of the mixture and the final properties of the hardened cement depend upon the composition (amount of cement, pozzolan, water) of the binder/stabilizer.

Ex situ S/S can be easily applied to excavated soils because methods are available to provide the vigorous mixing needed to combine the binder/stabilizer with the contaminated material. Pretreatment of the waste may be necessary to screen and crush large rocks and debris. Mixing can be performed via in-drum, in-plant or area mixing processes. In-drum mixing may be preferred for treatment of small volumes of waste or for toxic wastes. In-plant processes utilize rotary drum mixers for batch processes or pug mill mixers for continuous treatment. Larger volumes of waste may be excavated and moved to a contained area for area mixing. This process involves layering the contaminated material with the stabilizer/binder, and subsequent mixing with a backhoe or similar equipment. Mobile and fixed treatment plants are available for ex situ S/S treatment. Smaller pilot-scale plants can treat up to 100 tons of contaminated soil per day, while larger portable plants typically process 500 to over 1000 tons per day (Smith et al., 1995).

S/S techniques are available to provide mixing of the binder/stabilizer with the contaminated soil *in situ*. *In situ* S/S is less labor and energy intensive than ex situ process that require excavation, transport and disposal of the treated material. *In situ* S/S is also preferred if volatile or semi volatile organics are present because excavation would expose these contaminants to the air (U.S. EPA, 1990a). However the presence of bedrock, large boulders,

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cohesive soils, oily sands and clays may preclude the application of *in situ* S/S at some sites. It is also more difficult to provide uniform and complete mixing through *in situ* processes.

Mixing of the binder and contaminated matrix may be achieved using in-place mixing, vertical auger mixing or injection grouting. In-place mixing is similar to ex situ area mixing except that the soil is not excavated prior to treatment. The *in situ* process is useful for treating surface or shallow contamination and involves spreading and mixing the binders with the waste using conventional excavation equipment such as draglines, backhoes or clamshell buckets. Vertical auger mixing uses a system of augers to inject and mix the binding reagents with the waste. Larger (6-12 ft diameter) augers are used for shallow (10-40 ft) drilling and can treat 500-1000 cubic yards per day (Ryan and Walker, 1992; Jasperse and Ryan, 1992). Deep stabilization/solidification (up to 150 ft) can be achieved by using ganged augers (up to 3 ft in diameter each) that can treat 150-400 cubic yards per day. Finally injection grouting may be performed to inject the binder containing suspended or dissolved reagents into the treatment area under pressure. The binder permeates the surrounding soil and cures in place (Smith et al., 1995).

### **3.2.2.2 Vitrification**

The mobility of metal contaminants can be decreased by high-temperature treatment of the contaminated area that results in the formation of vitreous material, usually an oxide solid. During this process, the increased temperature may also volatilize and/or destroy organic contaminants or volatile metal species (such as Hg) that must be collected for treatment or disposal. Most soils can be treated by vitrification and a wide variety of inorganic and organic contaminants can be targeted. Vitrification may be performed *ex situ* or *in situ*, although *in situ* processes are preferred due to the lower energy requirements and cost (U.S. EPA, 1992a).

Typical stages in *ex situ* vitrification processes may include excavation, pretreatment, mixing, feeding, melting and vitrification, off-gas collection and treatment, and forming or casting of the melted product. The energy requirement for melting is the primary factor influencing the cost of *ex situ* vitrification. Different sources of energy can be used for this purpose, depending on local energy costs. Process heat losses and water content of the feed should be controlled in order to minimize energy requirements. Vitrified material with certain characteristics may be obtained by using additives such as sand, clay and/or native soil. The vitrified waste may be recycled and used as clean fill, aggregate, or other reusable materials (Smith et al., 1995).

*In situ* vitrification (ISV) involves passing electric current through the soil using an array of electrodes inserted vertically into the contaminated region. Each setting of four electrodes is referred to as a melt. If the soil is too dry, it may not provide sufficient conductance and a trench containing flaked graphite and glass frit (ground glass particles) must be placed between the electrodes to provide an initial flow path for the current. Resistance heating in

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the starter path melts the soil. The melt grows outward and down as the molten soil usually provides additional conductance for the current. A single melt can treat up to 1000 tons of contaminated soil to depths of 20 feet, at a typical treatment rate of 3 to 6 tons per hour. Larger areas are treated by fusing together multiple individual vitrification zones. The main requirement for *in situ* vitrification is the ability of the soil melt to carry current and solidify as it cools. If the alkali content (as Na<sub>2</sub>O and K<sub>2</sub>O) of the soil is too high ( $\geq 1.4$  wt%) the molten soil may not provide enough conductance to carry the current (Buelt and Thompson, 1992).

### 3.2.3 Toxicity and/or Mobility Reduction

Chemical and/or biological processes can be used to alter the form of metal contaminants in order to decrease their toxicity and/or mobility.

#### 3.2.3.1 Chemical Treatment

Chemical reactions can be initiated that are designed to decrease the toxicity or mobility of metal contaminants. The three types of reactions that can be used for this purpose are oxidation, reduction, and neutralization reactions. Chemical oxidation changes the oxidation state of the metal atom through the loss of electrons. Commercial oxidizing agents are available for chemical treatment, including potassium permanganate, hydrogen peroxide, hypochlorite and chlorine gas. Reduction reactions change the oxidation state of metals by adding electrons. Commercially available reduction reagents include alkali metals (Na, K), sulfur dioxide, sulfite salts, and ferrous sulfate. Changing the oxidation state of metals by oxidation or reduction can detoxify, precipitate, or solubilize the metals (NRC, 1994). Chemical neutralization is used to adjust the pH balance of extremely acidic or basic soils and/or groundwater. This procedure can be used to precipitate insoluble metal salts from contaminated water, or in preparation for chemical oxidation or reduction.

Chemical treatment can be performed *ex situ* or *in situ*. However *in situ* chemical agents must be carefully selected so that they do not further contaminate the treatment area. The primary problem associated with chemical treatment is the nonspecific nature of the chemical reagents. Oxidizing/reducing agents added to the matrix to treat one metal will also target other reactive metals and can make them more toxic or mobile (NRC, 1994). Also, the long-term stability of reaction products is of concern since changes in soil and water chemistry might reverse the selected reactions.

Chemical treatment is often used as pretreatment for S/S and other treatment technologies. Reduction of Cr(VI) to Cr(III) is the most common form of chemical treatment and is necessary for remediation of wastes containing Cr(VI) by precipitation or S/S. Chromium in its Cr(III) form is readily precipitated by hydroxide over a wide range of pH values. Acidification may also be used to aid in Cr(VI) reduction. Arsenic may be treatable by chemical oxidation since arsenate, As(V), is less toxic, soluble and mobile than arsenite, As(III). Bench-scale work has indicated that arsenic stabilization may be achieved by precipitation and coprecipitation with Fe(III) (Smith et al., 1995).

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### 3.2.3.2 Permeable Treatment Walls

Treatment walls remove contaminants from groundwater by degrading, transforming, precipitating or adsorbing the target solutes as the water flows through permeable trenches containing reactive material within the subsurface (Vidic and Pohland, 1996). Several methods are available for installation of permeable treatment walls, some of which employ slurry wall construction technology to create a permeable reactive curtain. The reactive zone can use physical, chemical and biological processes, or a combination of these. The ground-water flow through the wall may be enhanced by inducing a hydraulic gradient in the direction of the treatment zone or channeling ground-water flow toward the treatment zone (NRC, 1994).

Several types of treatment walls are being tried for arresting transport of metals in groundwater at contaminated sites. Trench materials being investigated include zeolite, hydroxyapatite, elemental iron, and limestone (Vidic and Pohland, 1996). Applications of elemental iron for chromium (VI) reduction and limestone for lead precipitation and adsorption are described below.

#### Elemental Iron

Trenches filled with elemental iron have shown promise for remediation of metals-contaminated sites. While investigations of this technology have focused largely on treatment of halogenated organic compounds, studies are being performed to assess the applicability to remediation of inorganic contaminants (Powell et al., 1994).

Low oxidation-state chemical species can serve as electron donors for the reduction of higher oxidation-state contaminants. This ability can be exploited to remediate metals that are more toxic and mobile in higher oxidation states, such as Cr(VI). Results of column experiments performed by Powell et al. (1994) and batch experiments performed by Cantrell et al. (1995) showed that chromate reduction was enhanced in systems containing iron filings in addition to the natural aquifer material. A field experiment has been initiated by researchers at the U.S. EPA National Risk Management Research Laboratory to investigate the use of zero-valent iron for chromium remediation at the U.S. Coast Guard air support base near Elizabeth City, North Carolina. Preliminary results indicate that the test barrier has reduced chromate in the groundwater to below detection limits (Wilson, 1995).

#### Limestone Barriers

The use of limestone treatment walls has been proposed for sites with metals contamination, in particular former lead acid battery recycling sites which have lead and acid contamination in groundwater and soil. In such cases, a limestone trench can provide neutralization of acidic groundwater. The attendant rise in pH promotes immobilization of any dissolved lead through precipitation and/or adsorption onto minerals. A limestone trench system is in design for implementation at the Tonolli Superfund site in Nesquehoning, Pennsylvania (U.S. EPA, 1992b)

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There is some experience in the coal mining industry with use of limestone in the manner anticipated for the Tonolli site. Most of this experience has been acquired since 1990, when the concept of “anoxic limestone drains” was introduced (Turner and McCoy, 1990). Since that time, numerous limestone drain systems have been installed at Appalachian coal field sites (primarily in Kentucky, West Virginia, and Pennsylvania) in an attempt to control acid mine drainage. Summaries of installations and evolving design considerations are provided in Hedin and Nairn (1992), Hedin et al. (1994), and Hedin and Watzlaf (1994).

Design and operating guidelines for the anoxic limestone drains have for the most part been developed from trial and observation. Briefly, the systems in use employ fairly large, #3 or #4 (baseball size) limestone rocks. Anoxic mine water is directed to the limestone drain, which is installed with a soil cover to inhibit contact with air. Hedin and Nairn (1992) report that “some systems constructed with limestone powder and gravel have failed, apparently because of plugging problems.” Preliminary review of the literature on design of anoxic limestone drains indicates primary concern with maintenance of anoxic conditions in the drains. If high dissolved concentrations of Fe are present and aerobic conditions develop, insoluble ferric hydroxide can form and coat the limestone, rendering it ineffective. High concentrations of aluminum are also a concern, as aluminum hydroxide can precipitate and yield the same kind of coating problems. With use of large diameter stones, plugging is prevented even if precipitation occurs and the stones become coated with precipitate.

Available operating data for anoxic limestone drains indicate that they can be effective in raising the pH of strongly acidic water. Hedin and Watzlaf (1994) reviewed operating data for 21 limestone drain systems. The data they compiled showed fairly consistent increases in pH of highly acidic mine drainage (at pH 2.3 to 3.5) to pH values in the range of 6.0 to 6.7. Thus, there is clearly precedent for employing the limestone drain approach with some confidence of success in raising pH of highly acidic water. Long term (i.e., greater than 10 years) performance cannot be predicted with confidence as there has been relatively short duration operating experience. However, experience to date indicates clearly that limestone drain systems can operate effectively under appropriate conditions, especially anoxic or low-oxygen groundwater, for at least several years.

### **3.2.3.3 Biological Treatment**

Biological treatment technologies are available for remediation of metals-contaminated sites. These technologies are commonly used for the remediation of organic contaminants and are beginning to be applied for metal remediation, although most applications to date have been at the bench and pilot scale (Schnoor, 1997). Biological treatment exploits natural biological processes that allow certain plants and microorganisms to aid in the remediation of metals. These processes occur through a variety of mechanisms, including adsorption, oxidation and reduction reactions, and methylation ( Means and Hincsee, 1994).

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## Bioaccumulation

Bioaccumulation involves the uptake of metals from contaminated media by living organisms or dead, inactive biomass. Active plants and microorganisms accumulate metals as the result of normal metabolic processes via ion exchange at the cell walls, complexation reactions at the cell walls, or intra- and extracellular precipitation and complexation reactions. Adsorption to ionic groups on the cell surface is the primary mechanism for metal adsorption by inactive biomass. Accumulation in biomass has been shown to be as effective as some ion exchange resins for metals removal from water (Means and Hinchee, 1994).

## Phytoremediation

Phytoremediation refers to the specific ability of plants to aid in metal remediation. Some plants have developed the ability to remove ions selectively from the soil to regulate the uptake and distribution of metals. Most metal uptake occurs in the root system, usually via absorption, where many mechanisms are available to prevent metal toxicity due to high concentration of metals in the soil and water. Potentially useful phytoremediation technologies for remediation of metals-contaminated sites include phytoextraction, phytostabilization and rhizofiltration (U.S. EPA, 1996b).

### *Phytoextraction*

Phytoextraction employs hyperaccumulating plants to remove metals from the soil by absorption into the roots and shoots of the plant. A hyperaccumulator is defined as a plant with the ability to yield  $\geq 0.1\%$  chromium, cobalt, copper or nickel or  $\geq 1\%$  zinc, manganese in the aboveground shoots on a dry weight basis. The aboveground shoots can be harvested to remove metals from the site and subsequently disposed as hazardous waste or treated for the recovery of the metals.

### *Phytostabilization*

Phytostabilization involves the use of plants to limit the mobility and bioavailability of metals in soil. Phytostabilizers are characterized by high tolerance of metals in surrounding soils but low accumulation of metals in the plant. This technique may be used as an interim containment strategy until other remediation techniques can be developed, or as treatment at sites where other methods would not be economically feasible.

### *Rhizofiltration*

Rhizofiltration removes metals from contaminated groundwater via absorption, concentration and precipitation by plant roots. This technique is used to treat contaminated water rather than soil and is most effective for large volumes of water with low levels of metal contamination. Terrestrial plants are more effective than aquatic plants because they develop a longer, more fibrous root system that provides a larger surface area for interaction.

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Wetlands construction is a form of rhizofiltration that has been demonstrated as a cost-effective treatment for metals-contaminated wastewater.

### Bioleaching

Bioleaching uses microorganisms to solubilize metal contaminants either by direct action of the bacteria, as a result of interactions with metabolic products, or both. Bioleaching can be used *in situ* or *ex situ* to aid the removal of metals from soils. This process is being adapted from the mining industry for use in metals remediation. The mechanisms responsible for bioleaching are not fully defined, but in the case of mercury bioreduction (to elemental mercury) is thought to be responsible for mobilization of mercury salts (Means and Hincsee, 1994).

### Biochemical Processes

Microbially mediated oxidation and reduction reactions can be manipulated for metal remediation. Some microorganisms can oxidize/reduce metal contaminants directly while others produce chemical oxidizing/reducing agents that interact with the metals to effect a change in oxidation state. Mercury and cadmium have been observed to be oxidized through microbial processes, and arsenic and iron are readily reduced in the presence of appropriate microorganisms. The mobility of metal contaminants is influenced by their oxidation state. Redox reactions can therefore be used to increase or decrease metal mobility (Means and Hincsee, 1994).

Methylation involves attaching methyl groups to inorganic forms of metal ions to form organometallic compounds. Methylation reactions can be microbially mediated. Organometallic compounds are more volatile than inorganic metals and this process can be used to remove metals through volatilization and subsequent removal from the gas stream. However, organometallics are also more toxic and mobile than other metal forms and may potentially contaminate surrounding surface waters and groundwater (Means and Hincsee, 1994).

### **3.2.4 Physical Separation**

Physical separation is an *ex situ* process that attempts to separate the contaminated material from the rest of the soil matrix by exploiting certain characteristics of the metal and soil. Physical separation techniques are available that operate based on particle size, particle density, surface and magnetic properties of the contaminated soil. These techniques are most effective when the metal is either in the form of discrete particles in the soil or if the metal is sorbed to soil particles that occur in a particular size fraction of the soil. Physical separation is often used as a form of pretreatment in order to reduce the amount of material requiring subsequent treatment (Rosetti, 1993). Several techniques are available for physical separation of contaminated soils including screening, classification, gravity concentration, magnetic separation and froth flotation.

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Screening separates soils according to particle size by passing the matrix through a sieve with particular size openings. Smaller particles pass through the sieve and leave larger particles behind, however, the separation is not always complete. Screening may be performed as a stationary process or with motion using a wet or dry process stream (Smith et al., 1995).

Classification involves separation of particles based upon the velocity with which they fall through water (hydroclassification) or air (air classification). Hydroclassification is more common for soil separation and may be performed using a non-mechanical, mechanical or a hydraulic classifier (Rosetti, 1993).

Gravity concentration relies on gravity and one or more other forces (centrifugal force, velocity gradients, etc.) that may be applied to separate particles on the basis of density differences. Gravity concentration may be achieved through the use of a hydrocyclone, jig, spiral concentrator, or shaking table (Rosetti, 1993).

Froth flotation uses air flotation columns or cells to remove particles from water. In this process, air is sparged from the bottom of a tank or column that contains a slurry of the contaminated material. Some metals and minerals attach to the air bubbles due to particular surface properties, such as hydrophobicity. Froth flotation can be used to remove metals that attach to air bubbles, or to remove other minerals while the metal remains in the slurry (Rosetti, 1993).

Magnetic separation subjects particles to a strong magnetic field using electromagnets or magnetic filters and relies on differences in magnetic properties of minerals for separation. Low intensity wet magnetic separators are the most common magnetic separation devices. This process can recover a wide variety of minerals and is particularly successful for separating ferrous from nonferrous minerals (Allen and Torres, 1991).

### **3.2.5 Extraction**

Metals-contaminated sites can be remediated using techniques designed to extract the contaminated fraction from the rest of the soil, either *in situ* or *ex situ*. Metal extraction can be achieved by contacting the contaminated soil with a solution containing extracting agents (soil washing and *in situ* soil flushing) or by electrokinetic processes. The contaminated fraction of soil and/or process water is separated from the remaining soil and disposed or treated.

#### **3.2.5.1 Soil Washing**

Soil washing can be used to remove metals from the soil by chemical or physical treatment methods in aqueous suspension. Soil washing is an *ex situ* process that requires soil excavation prior to treatment. Chemical treatment involves addition of extraction agents that react with the contaminant and leach it from the soil (Elliot and Brown, 1989; Ellis and

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Fogg, 1985; Tuin and Tels, 1990). The liquid containing the contaminants is separated from the soil resulting in a clean solid phase. Physical treatment is achieved by particle size separation technologies adapted from mineral processing to concentrate the contaminant in a particular size fraction (Allen and Torres, 1991).

Fine particles (<63  $\mu\text{m}$ ) often contain the majority of contaminated material because they bind contaminants strongly due to their large and reactive surface area. Many current soil washing approaches attempt to separate the fine fraction from the remainder of the soil in order to reduce the amount of material for subsequent treatment or disposal (Rosetti, 1993). Particle size separation techniques may not be successful if fine particle, e.g., metal oxide, coatings are present on particles in larger size fractions (Van Ben Schoten et al., 1994).

### Preliminary Screening

After excavation, the soil undergoes preliminary screening and preparation in order to separate large rocks and debris from the contaminated matrix. Residual fines may be adhered to the surface of large rocks and are often washed off prior to return of the large rocks to the site (Rosetti, 1993).

### Secondary Screening

Most soil washing processes employ secondary screening to segregate the particles into different size fractions, usually between 5 mm and 60 mm. Most secondary screening processes involve making an aqueous slurry of the soil stream and wet screening/sieving of the slurry. The particles in this size range are considered less contaminated than the finer fraction and may be returned to the site as clean soil after separation from the water (Rosetti, 1993).

### Chemical Treatment

Chemical treatment may be used to solubilize contaminants from the most contaminated fraction of the soil. Chemical treatment is performed in an aqueous slurry of the contaminated material to which an extracting agent is added. The extraction is performed in a mixing vessel or in combination with the physical treatment stage. The type of extractant used will depend on the contaminants present and the characteristics of the soil matrix. Many processes manipulate the acid/base chemistry of the slurry to leach contaminants from the soil (Tuin and Tels, 1990). However, if a very low pH is required concerns about dissolution of the soil matrix may arise. Chelating agents (e.g., EDTA) selectively bind with some metals and may be used to solubilize contaminants from the soil matrix (Elliot and Brown, 1989). Oxidizing and reducing agents (e.g., hydrogen peroxide, sodium borohydride) provide yet another option to aid in solubilization of metals since chemical oxidation/reduction can convert metals to more soluble forms (Assink and Rulkens, 1989; Tuin et al., 1987). Finally, surfactants may be used in extraction of metals from soil (U.S. EPA, 1996b).

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## Physical Treatment

Physical treatment is used to separate the contaminated fraction, usually the fine materials, from the rest of the soil matrix. Physical separation may be performed alone (Section 3.2.4) or in conjunction with chemical treatment, as in most soil washing processes. The most common method for physical separation in soil washing uses rotary attrition scrubbers to isolate the contaminated particles. The rotation of the slurry causes contact between large particles, resulting in attrition of the larger particles which releases the contaminant and contaminated fines to the slurry. The contaminant remains suspended in solution or sorbs to the reactive fine particles. Vibration units are also available to perform similar separations (Rosetti, 1993).

Hydrocyclones are the most common method used to separate fines from the clean soil. Other options are available for fine particle separation, including mechanical classifiers, gravity classifiers, spiral concentrators, and magnetic separators (Rosetti, 1993).

Froth flotation can be used to combine physical and chemical treatment processes into one step. For this method, extracting agent is added to the soil before it enters the froth flotation cell. The slurry is leached in the tanks to remove the contaminant and the fines (<50 µm) are then separated from coarse particles in the flotation unit (Rosetti, 1993).

## Dewatering

After the contaminated fine particles are separated from the clean coarse particles, both fractions are dewatered. The fine fraction is usually dewatered using a belt filter or filter press and disposed of in a landfill. Larger particles are rinsed to remove residual extracting solution and contaminant and dewatered using belt and filter presses. This fraction is considered clean and can be returned to the site.

## Water Treatment

The contaminated water from rinsing and dewatering steps is treated by manipulating the solution chemistry to separate the contaminant from the extractant if possible. Contaminants can then be removed from solution, most commonly by precipitation or sedimentation, and are dewatered before disposal with the contaminated fines. The extracting agent and process water can be recycled for reuse.

### **3.2.5.2 Pyrometallurgical Extraction**

Pyrometallurgical technologies use elevated temperature extraction and processing for removal of metals from contaminated soils. Soils are treated in a high-temperature furnace to remove volatile metals from the solid phase. Subsequent treatment steps may include metal recovery or immobilization. Pyrometallurgical treatment requires a uniform feed material for efficient heat transfer between the gas and solid phases and minimization of

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particulates in the off-gas. This process is usually preceded by physical treatment to provide optimum particle size. Pyrometallurgical processes usually produce a metal-bearing waste slag, but the metals can also be recovered for reuse (U.S. EPA, 1996c).

### **3.2.5.3 *In Situ Soil Flushing***

*In situ* soil flushing is used to mobilize metals by leaching contaminants from soils so that they can be extracted without excavating the contaminated materials. An aqueous extracting solution is injected into or sprayed onto the contaminated area to mobilize the contaminants usually by solubilization. The extractant can be applied by surface flooding, sprinklers, leach fields, vertical or horizontal injection wells, basin infiltration systems or trench infiltration systems (U.S. EPA, 1996b). After being contacted with the contaminated material the extractant solution is collected using pump-and-treat methods for disposal or treatment and reuse. Similar extracting agents are used for *in situ* soil flushing and soil washing, including acids/bases, chelating agents, oxidizing/reducing agents and surfactants/cosolvents. Also, water can be used alone to remove water-soluble contaminants such as hexavalent chromium. The applicability of *in situ* soil flushing technologies to contaminated sites will depend largely on site-specific properties, such as hydraulic conductivity, that influence the ability to contact the extractant with contaminants and to effectively recover the flushing solution with collection wells (NRC,1994).

### **3.2.5.4 *Electrokinetic Treatment***

Electrokinetic remediation technologies apply a low density current to contaminated soil in order to mobilize contaminants in the form of charged species. The current is applied by inserting electrodes into the subsurface and relying on the natural conductivity of the soil (due to water and salts) to effect movement of water, ions and particulates through the soil. Water and/or chemical solutions can also be added to enhance the recovery of metals by this process. Positively charged metal ions migrate to the negatively charged electrode, while metal anions migrate to the positively charged electrode. Electrokinetic treatment concentrates contaminants in the solution around the electrodes. The contaminants are removed from this solution by a variety of processes, including electroplating at the electrodes, precipitation/coprecipitation at the electrodes, complexation with ion exchange resins, or by pumping the water from the subsurface and treating it to recover the extracted metals (Smith et al, 1995).

Electrokinetic treatment is most applicable to saturated soils with low ground-water flow rates and moderate to low permeability. The efficiency of metal removal by this process will be influenced by the type and concentration of contaminant, the type of soil, soil structure, and interfacial chemistry of the soil.

## **3.3 PERFORMANCE OF AVAILABLE COMMERCIAL TECHNOLOGIES**

The following section focuses on commercially available technologies that have been demonstrated or implemented for metals-contaminated soils and groundwater. Ex situ

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treatment technologies are examined only for soils. The full range of contaminated water treatment technologies is available for ex situ treatment of groundwater. For the most part, the technologies reported are those whose performance has been verified by the U.S. EPA under the Superfund Innovative Technology Evaluation (SITE) program which evaluates emerging and demonstrated technologies. Technologies currently in the SITE demonstration phase are also discussed.

### **3.3.1 Superfund Innovative Technology Evaluation (SITE) Demonstration and Best Demonstrated Available Technology (BDAT) Status**

The 1986 Superfund Amendments and Reauthorization Act recognized a need for an “alternative or innovative technology research and demonstration program.” In response, the U.S. EPA established the Superfund Innovative Technology Evaluation (SITE) Program to encourage the development and implementation of innovative treatment technologies for remediation of hazardous waste sites and for monitoring and measurement. Innovative technologies are field-tested in the SITE Demonstration Program and engineering and cost data are collected to assess the performance of the technology. The demonstration stage also attempts to evaluate the applicability of the technology to different types of wastes and waste matrices, the need for pre- and post-processing of the waste stream, and potential operating problems. The SITE Program is administered by the U.S. EPA Office of Research and Development (ORD) National Risk Management Research Laboratory, headquartered in Cincinnati, Ohio.

The Resource Conservation and Recovery Act (RCRA) provides for determination of a Best Demonstrated Available Technology (BDAT) for treatment of hazardous wastes. BDATs have been established based upon critical analysis of performance data collected for treatment of various industry-generated wastes. BDAT status is given only to proven, commercially available technologies. Different BDATs and treatment standards are usually given for nonwastewater and wastewater forms of contamination. The applicability of a BDAT to metals-contaminated soil and water at a Superfund site must be evaluated on a site specific basis. The establishment of a BDAT does not prevent the use of other available technologies for treatment of these wastes.

### **3.3.2 Containment**

Containment technologies are widely used to control the transport of hazardous materials and prevent the spread of contamination. Containment is the preferred remedial method for sites having low levels of wastes with low toxicity and low mobility, or wastes that have been pretreated to obtain these characteristics. Containment may also be used as a temporary measure to reduce the mobility of wastes that pose a high risk until a permanent remedy is selected and implemented. Advantages to containment technologies include relatively simple and rapid implementation often at lower cost than alternatives that require excavation; ability to treat large areas and volumes of waste; and the potential for successful containment as the final action at the site. Uncertainty regarding long-term effectiveness

and the need for long-term inspection because untreated contaminants remain onsite are among the disadvantages of containment technologies. Also, future use of the site may be limited if containment technologies are used. Containment has been selected as the remedial operation for soil contaminated with metals at a number of sites. Some example applications are summarized in Table 2.

**Table 2. Example Containment Applications at Metals-Contaminated Superfund Sites (from U.S. EPA, 1996c)**

Site Name/State	Containment Technology	Metal Contaminants	Secondary Technology	Status <sup>a</sup>
Ninth Avenue Dump, IN	Slurry Wall	Pb	Capping	S
Industrial Waste Control, AK	Slurry Wall	As, Cd, Cr, Pb	Capping, French Drain	I
E.H. Shilling Landfill, OH	Slurry Wall	As	Capping, Clay Berm	S
Chemtronic, NC	Capping	Cr, Pb		S
Ordnance Works Disposal, WV	Capping	As, Pb		S
Industriplex, MA	Capping	As, Pb, Cr		I

<sup>a</sup> Status codes as of February, 1996: I=in operation; S=selected

Capping systems have been selected for a number of sites with low levels of metal contamination. Monitoring wells and/or infiltration monitoring systems are often used to help assess the performance of capping systems.

Slurry walls have also been used for containment of metals-contaminated sites. The performance of vertical containment barriers also must be monitored. Performance can be influenced by geography, topography, and geology. The presence of certain compounds can also influence the long-term integrity of some cement-based vertical barriers by chemically attacking the soil-bentonite blends. Material availability can affect the application of slurry walls and other containment technologies.

There are no established BDAT's for containment technologies since they are not considered to be treatment technologies. Ongoing SITE demonstrations for remediation of metals by containment technologies include a high clay grouting procedure (Morrison Knudsen Corporation) and frozen soil barriers (RKK, Ltd.).

### 3.3.3 Ex Situ Remediation

The majority of the technologies that have been demonstrated for metals remediation to date are ex situ technologies. Ex situ remediation technologies demonstrated include solidification/stabilization, soil washing, vitrification and pyrometallurgical separation.

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### **3.3.3.1 Solidification/Stabilization**

Immobilization technologies, especially solidification/stabilization, are the most common methods selected for remediation of metal contamination, accounting for nearly 30 percent of all soil treatment technologies at Superfund sites. S/S techniques have been widely used to manage metal wastes at hazardous waste sites and to treat residues from other treatment processes (LaGrega et al., 1994). Benefits associated with immobilization treatments include their broad application to a wide variety of metals (Malone and Jones, 1985) and also to wastes that contain mixtures of metals and organics (U.S. EPA, 1996b).

Solidification/stabilization technologies using cement-based and pozzolan binders are available commercially and have been applied at several sites for a wide variety of metals, including chromium, lead, arsenic, mercury, and cadmium (Lo et al., 1988; Stanczyk and Senefelder, 1982; Zirschky and Piznar, 1988; Lister, 1996).

Examples of sites where ex situ S/S technologies have been selected and/or implemented for remediation of metals-contaminated soils are given in Table 3. Remediation has been completed for a number of these sites and S/S has been selected or initiated for several others. SITE demonstrations have been performed or are underway for various ex situ stabilization/solidification technologies.

**Table 3. Example Solidification/Stabilization Applications at Selected Metals-Contaminated Superfund Sites (from U.S. EPA, 1996c)**

Site Name/State	S/S Technology	Metal Contaminants	Secondary Technology	Status <sup>a</sup>
DeRewal Chemical, NJ	Solidification	Cr, Cd, Pb	GW pump and treat	S
Marathon Battery Co., NY	Stabilization	Cd, Ni	Dredging, off-site disposal	I
Nascolite, NJ	Stabilization	Pb	On-site disposal of stabilized soil, off-site disposal of wetland soil	S
Roebling Steel, NJ	S/S	As, Cr, Pb	Capping	S
Waldick Aerospace, NJ	S/S	Cd, Cr	Off-site disposal	C
Aladdin Plating, PA	Stabilization	Cr	Off-site disposal	C
Palmerton Zinc, PA	Stabilization	Cd, Pb	-	I
Tonolli Corp.	S/S	As, Pb	<i>In situ</i> chemical barrier	S
Whitmoyer Laboratories, PA	Oxidation/Stabilization	As	GW pump and treat, capping, grading, revegetation	S
Bypass 601, NC	S/S	Cr, Pb	GW pump and treat, capping, grading, revegetation	S
Flowood, MS	S/S	Pb	Capping	C
Independent Nail, SC	S/S	Cd, Cr	Capping	C
Pepper's Steel and Alloys, FL	S/S	As, Pb	On-site disposal	C
Pesses Chemical, TX	Stabilization	Cd	Capping	C
E.I. Dupont de Nemours, IA	S/S	Cd, Cr, Pb	Capping, regrading, revegetation	C
Shaw Avenue Dump, IA	S/S	As, Cd	Capping, GW monitoring	C
Frontier Hard Chrome, WA	Stabilization	Cr	-	S
Gould Site, OR	S/S	Pb	Capping, regrading, revegetation	I

<sup>a</sup> Status codes as of February, 1996: C=completed; I=in operation; S=selected

Ex situ solidification/stabilization techniques have been determined to be the BDAT for a range of waste types, including cadmium nonwastewaters (other than Cd-containing batteries), chromium nonwastewaters (after reduction to Cr(III)), lead nonwastewaters, wastes with low (<260mg/kg) concentrations of elemental mercury, and plating and steel-making wastes. S/S can also be used to treat arsenic wastes even though vitrification was selected as the BDAT for arsenic-containing nonwastewaters (U.S. EPA, 1996c).

### 3.3.3.2 Soil Washing

Soil washing technologies are applicable to a range of soils containing a variety of metal contaminants. Soil washing is most easily implemented when a single metal contaminant occurs in a particular insoluble fraction of the soil which can be separated by particle size

classification. Soils with a minimum content of finer material (<20% of particles with diameters <2 mm) are easier to process. Soil washing has been used for remediation of metals-contaminated sites in Europe and has been selected and/or implemented at several U.S. Superfund sites. Table 4 gives examples of Superfund sites at which soil washing has been selected as the remediation technology.

**Table 4. Example Soil Washing Applications at Metals-Contaminated Superfund Sites (from U.S. EPA, 1996c)**

Site Name/State	Soil Washing Technology	Metal Contaminants	Secondary Technology	Status <sup>a</sup>
Ewan Property, NJ	Water washing	As, Cr, Cu, Pb	Solvent extraction to remove organics	S
GE Wiring Devices, PR	Water with KI solution additive	Hg	On-site disposal of clean soil	S
King of Prussia, NJ	Water with washing agents	Ag, Cr, Cu	Sludge disposal	C
Zanesville Well Field, OH	Water washing	As, Cr, Hg, Pb	On-site disposal of clean soil, SVE to remove organics	S
Twin Cities Army Ammunition Plant, MN	Acid leaching	Cd, Cr, Cu, Hg, Pb	Soil leaching	C
Sacramento Army Depot, CA	Water washing	As, Cr, Pb	Off-site treatment/disposal of wash liquid, on-site disposal of clean soil	S/D

<sup>a</sup>Status codes as of February, 1996: C=completed; S=selected; S/D=selected but subsequently deselected

Remediation at the Twin Cities Army Ammunition Plant (TCAAP) in New Brighton, Minnesota is one of the two completed soil washing projects. The COGNIS TERRAMET® soil washing procedure used at this site employed a combination of particle sizing, gravity separation, and acid-leaching apparatus that was designed to remove lead, mercury, cadmium, chromium and copper from the soil. Preliminary studies have shown that the primary target metal at this site, lead, could be reduced from over 86000 mg/kg to less than 100 mg/kg, well below the target cleanup level of 300 mg/kg (Griffiths, 1995). Acid leaching soil washing procedures have been designated as the BDAT for mercury-contaminated soils. Several SITE demonstrations have been performed for soil washing of metals-contaminated soils (U.S. EPA, 1996c).

### 3.3.3.3 Vitrification

Vitrification is most applicable to sites containing low-volatility metals with high glass solubilities, and therefore appears to be well-suited for treatment of lead, chromium, arsenic,

zinc, cadmium and copper wastes (Table 5). The ability of a melt to retain these and other metals depends on the metal solubility in the soil at the site, and silica content of the soil. The metal concentration can be adjusted by adding soil or another source of silica to improve site characteristics for vitrification. The ability to control volatile emissions also influences the applicability of vitrification technologies. Mercury's high volatility and low glass solubility makes it unsuitable for vitrification, but treatment by vitrification may be allowed at sites containing very low mercury concentrations.

Ex situ vitrification may not be applicable for soils with greater than 25% moisture content due to excess fuel consumption, or at sites where size reduction and classification are not feasible. Several ex situ vitrification technologies are under development. SITE program demonstrations have been completed for two of these processes and a third demonstration is underway (U.S. EPA, 1996c). *Ex-situ* vitrification has also been demonstrated for treatment of cesium-contaminated tank wastes from the Oak Ridge Reservation (PNNL, 1997).

**Table 5. Approximate Vapor Pressures and Glass Solubility Limits for Metals**

Metal	Temperature [°C] at which Vapor Pressure=1mm Hg <sup>a</sup>	Maximum Allowed Oxide Content [%] for Sample Silicate Glass <sup>b</sup>
Pb	973	30
Cr	1840	2
As	372	5
Zn	487	20
Cd	394	1
Cu	1628	5
Hg	126.2	~0

<sup>a</sup> CRC, 1991.

<sup>b</sup> From Smith et al., 1995

### 3.3.3.4 Pyrometallurgical Separation

Mercury has a relatively high vapor pressure and is easily converted to its metallic form at elevated temperature, making it easily treated by pyrometallurgic methods. Pyrometallurgic treatment of lead, arsenic, cadmium and chromium may require pretreatment by reducing agents or fluxing agents to facilitate melting. Nonvolatile metals such as chromium can be tapped from the furnace as molten metal (U.S. EPA, 1996c).

Pyrometallurgical treatment is usually performed offsite because few mobile treatment units are available. This technology is most applicable to large volumes of highly-contaminated soils (metal concentrations >5%-20%, especially when metal recovery is expected. Low metal concentrations can be processed, especially for mercury since it is easy to volatilize and recover (Smith et al., 1995).

A number of pyrometallurgical process technologies are currently available for treatment of metals-contaminated soils. Pyrometallurgical treatment is a BDAT for cadmium-containing batteries, lead nonwastewaters, mercury wastes, mercury from wastewater treatment sludge, lead acid batteries, and zinc nonwastewaters. SITE demonstrations have been completed for thermal desorption (RUST Remedial Services, Inc.) and flame reactor (Horsehead Resource Development Company, Inc.) pyrometallurgical technologies.

### 3.3.4 *In Situ* Remediation

*In situ* remediation technologies offer the potential for significant cost savings over *ex situ* technologies because *in situ* techniques are usually associated with lower labor and energy requirements for implementation. This section discusses the status of *in situ* technologies which are currently available for metal remediation at contaminated sites.

#### 3.3.4.1 Solidification/Stabilization

*In situ* S/S treatment appears to have been applied less frequently than *ex situ* techniques mostly due to concerns about uniformity of treatment and long-term reliability. These limitations are being reduced, however, through advances in chemical reagent delivery systems for large-diameter auger drilling devices (Jasperse, 1989; Walker, 1992). Examples of Superfund sites at which *in situ* S/S has been selected for remediation are given in Table 6. While *in situ* S/S technologies are well developed due to roots in construction techniques, data on the performance of *in situ* S/S are limited. Based upon preliminary data, *in situ* S/S appears likely to be an effective treatment option. *In situ* S/S typically will be most beneficial for sites with contamination at depths less than 8-10 feet and for larger volumes of waste because *ex situ* may prove to be cheaper for small volumes and shallow contamination due to high costs associated with mobilization and demobilization for *in situ* technologies. Deep soil mixing technology is also available for treating contaminated soils at greater depth (Ryan and Walker, 1992) but is more expensive than shallow soil mixing. The cost of *in situ* technologies is also affected by implementation concerns such as a level, stable base that is required for augering, and the presence of large rocks that can make large-diameter augering impossible. The use of dry reagents in soils with high moisture content is a well established method in Europe that is gaining interest in the U.S. and may expand the applicability of *in situ* S/S techniques (U.S. EPA, 1996b).

**Table 6. Example *In situ* Solidification/Stabilization Applications at Metals-Contaminated Superfund Sites (from U.S. EPA, 1996c)**

Site Name/State	S/S Technology	Metal Contaminants	Status <sup>a</sup>
Gurley Pit, AR	<i>In situ</i> S/S	Pb	C
General Electric Co., FL	<i>In situ</i> S/S	Pb, Cr, Cu, Zn	D

<sup>a</sup> Status codes as of February, 1996: C=completed; D=demonstrated

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### 3.3.4.2 Vitrification

*In situ* vitrification (ISV) technologies are currently offered commercially in the U.S. by a single vendor, Geosafe Corporation. The first full-scale application of ISV was demonstrated at the Parsons Chemical/ETM Enterprises Superfund site in Grand Ledge, Michigan under the EPA SITE program (Table 7). The Geosafe ISV process was used for treatment of soils and sediments contaminated with pesticides, metals (As, Cr, Hg, Pb), and dioxins. This treatment system required the use of eight melts which were each completed over a time frame of 10 to 20 days. This system also included an air emissions control system to treat volatilized contaminants, including mercury. While ISV is not recommended for remediation of mercury, this method can be used in conjunction with emissions control systems when Hg is present in mixed metal/organic wastes. This treatment was successful, meeting TCLP limits for all of the metals in the treated waste.

ISV has also been used successfully at two sites contaminated with organics (PCB, dioxin, pentachlorophenol, pesticides, herbicides), further demonstrating the applicability of this technology. Based upon observations from these limited applications, it appears that ISV may not be appropriate for sites with high levels of organics (> 10 % organics by weight) due to contamination of the off-gas, or inorganics (> 25 % metals by weight, or > 20 % by volume) due to concerns about exceeding glass solubility limits (U.S. EPA, 1996b).

**Table 7. Example *In Situ* Vitrification Applications at Metals-Contaminated Superfund Sites (from U.S. EPA, 1996c)**

Site Name/State	Vitrification Technology	Metal Contaminants	Status <sup>a</sup>
Parsons Chemical, MI	<i>In situ</i> vitrification	As, Cr, Hg, Pb	C
Rocky Mountain Arsenal, CO	<i>In situ</i> vitrification	As, Hg	S/D

<sup>a</sup> Status codes as of February, 1996: C=completed; S/D=selected but subsequently deselected

### 3.3.4.3 *In Situ* Soil Flushing

*In Situ* soil flushing has been selected for treatment at several Superfund sites contaminated with metals. Some examples of sites where *in situ* soil flushing is currently operational are given in Table 8. *In situ* soil flushing is the technology in design or the predesign stage at least five other sites. This technology has been applied for a limited number of projects, mostly containing organic forms of contamination (NRC, 1994), and limited information is available on the application of this technology to metals-contaminated sites.

**Table 8. Example *In Situ* Soil Flushing Applications at Metals-Contaminated Superfund Sites (from U.S. EPA, 1996c)**

Site Name/State	<i>In situ</i> soil flushing Technology	Metal Contaminants	Secondary Technology	Status <sup>a</sup>
Lipari Landfill, NJ	Flushing of contained wastes with water	Cr, Hg, Pb	Slurry wall, cap, excavation of wetlands	I
United Chrome Products, OR	Soil flushing with water	Cr	Considering electrokinetic and chemical (reduction) treatment	I

<sup>a</sup> Status codes as of February, 1996: I=in operation

The United Chrome Products Superfund site in Corvallis, Oregon is currently being remediated using *in situ* soil flushing technologies. The soil and groundwater at this site are heavily contaminated with chromium, with chromium levels in the soil as high as 60000 mg/kg and levels in the groundwater reaching up to 19000 mg/kg. The general approach to remediation of this site has been removal of the more soluble, mobile and toxic form of chromium, Cr(VI), by flushing the contaminated region with water to solubilize Cr(VI), with subsequent extraction of the chromium-containing water for treatment. Remediation at this site began in 1985 and has combined a variety of technologies to aid remediation by *in situ* soil flushing. The technologies used have included infiltration basins and trenches to flush contaminated soils, a 23-well ground-water extraction network to remove contaminated groundwater and recharge water, on-site treatment of wastewater, and off-site disposal of contaminated soil and debris (Sturges et al., 1992).

This full-scale application of *in situ* soil flushing with water as the flushing solution appears to be successful for removal of Cr(VI) from coarse soils of relatively high hydraulic conductivity. The *in situ* soil flushing procedure used at this site leaches contaminants from the unsaturated and saturated zones, and provides for recharge of the groundwater to the extraction wells. This cleanup operation has removed significant amounts of chromium from the soil and groundwater and the ground-water pumping strategy has achieved hydraulic containment of the plume. Cr(VI) levels in water retrieved by the extraction wells decreased from more than 5000 mg/L to approximately 50 mg/L during the first two and one half years of operation. Average chromium concentrations in the plume decreased from 1923 mg/L to 207 mg/L after flushing the first one and one half pore volumes (approximately 2.6 million gallons for one pore volume). These rapid removal rates are expected to continue for the first few pore volumes of treatment until Cr(VI) removal begins to tail off to the asymptotic level. Tailing results from slow desorption from soil particles, dissolution of solid phase contaminants, and release of contaminants from the fine pores in the soil matrix. Tailing is commonly observed in *in situ* soil flushing applications and usually represents the practical limit for remediation via pump and treat methods (Sturges et al., 1992).

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### 3.3.4.4 Electrokinetic Extraction

The success of various electrokinetic remediation technologies has been illustrated for removal of metals from soils via bench and pilot scale experiments. Currently, several of these technologies are being implemented in comprehensive demonstration studies to further the use of electrokinetic techniques at contaminated sites.

Electrokinetic remediation of metals-contaminated sites has been demonstrated *in situ* at many sites in Europe using processes developed by Geokinetics International, Inc. (GII) (U.S. EPA, 1996b). Table 9 provides examples of sites in Europe for which this technology has been selected as the remediation technology. The success of electrokinetic remediation appears to vary depending on the metals present, and can remove up to 90% of the initial contamination. The first demonstration of this electrokinetic process in the U.S. is scheduled under the EPA SITE program for remediation of a chromium-contaminated soil at the Sandia Chemical Waste Landfill.

**Table 9. Example Electrokinetic Applications at Metals-Contaminated Sites (from U.S. EPA, 1996c)**

Site Description	Electrokinetic Technology	Metal Contaminants	Status <sup>a</sup>
Former paint factory	Electrochemical Remediation	Cu, Pb	C
Operational galvanizing plant	Electrochemical Remediation	Zn	C
Former timber plant	Electrochemical Remediation	As	C
Temporary landfill	Electrochemical Remediation	Cd	C
Military air base	Electrochemical Remediation	Cd, Cr, Cu, Ni, Pb, Zn	C

<sup>a</sup> Status codes as of February, 1996: C=completed

Electrokinetics, Inc. is carrying out a SITE demonstration study of lead extraction from a creek bed at a U.S. Army firing range in Louisiana using their CADEX<sup>®</sup> electrode system. Soils at this site are contaminated with lead at concentrations up to 4500 mg/kg. In pilot-scale studies, the lead levels in the soil were reduced to below 300 mg/kg after 30 weeks of processing. The TCLP values for this soil were reduced from over 300 mg/L to less than 40 mg/L over this time. This technology is also being explored for remediation of sites contaminated with arsenic. Treatability and pilot-scale field testing studies for this application are under way.

Other electrokinetic techniques have been demonstrated for remediation of organics (TCE) and have accounted for removal of up to 98% of these wastes. The LASAGNA<sup>™</sup> process is being developed by a consortium consisting of Monsanto, E.I. DuPont deNemours & Co., Inc. and General Electric. LASAGNA<sup>™</sup> is an integrated, *in situ* process that uses

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electrokinetics to transport contaminants in soil pore water into treatment zones. The treatment zones are designed to capture or decompose the organic contaminants. ManTech Environmental provides the ElectroChemical GeoOxidation (ECGO) process that has been used to successfully remediate organic-contaminated soil and groundwater in Germany. ECGO uses induced electric currents to create oxidation-reduction reactions that mineralize organic contaminants. These reactions may be useful for immobilization of inorganic contaminants as well. Attempts are being made to determine the potential for treatment of metals using these processes (U.S. EPA, 1996b).

#### **3.3.4.5 Biological Treatment**

Phytoremediation technologies are largely in the developmental stage and many are being field tested at a variety of sites in the U.S. and in Europe. Because full-scale applications of phytoremediation technologies are just being initiated, limited cost and performance data are available. Some techniques under development have shown potential for use at metals-contaminated sites. Phytostabilization and phytoextraction methods are being developed by Phytotech, Inc. and field tests for patented phytoextraction techniques are being performed. Some grasses have been made commercially available for phytostabilization of metals (lead, copper, zinc) (Salt, 1995). Nickel has been removed from plating wastes by bacteria (Wong and Kwok, 1992) and other organisms are being genetically engineered to remove metals such as cadmium, cobalt, copper and mercury (Smit and Atwater, 1991). Bioreduction has been demonstrated (for Hg) at the bench scale but has not been tested at pilot scale (Smith et al., 1995). A process has been developed for chromium reduction by H<sub>2</sub>S produced by sulfate-reducing bacteria and reduction of Cr(VI) by direct metabolism is being investigated by several organizations (Smith et al., 1995).

Treatment by wetlands has been studied under the U.S. EPA's SITE program. Full-scale demonstration of a constructed wetland is planned for the Burleigh Tunnel site, part of the Clear Creek/Central City Superfund site in Colorado.

Bioleaching is currently used to recover copper and uranium ores by heap or *in situ* leaching (Ehrlich, 1988) and is under development for a wide range of metals including cadmium, chromium, lead, mercury and nickel. Microorganisms have been tested for chemical reduction and removal of mercury salts from wastewater (Horn et al. 1992, Hansen and Stevens, 1992).

Phytoremediation technologies will likely be limited to use in shallow soils with relatively low levels of metal contamination. Based upon estimates of biomass productivity and metal content of soils, the annual removal rate of metals by phytoremediation would be limited to between 2.5 to 100 mg/kg of soil contaminants (U.S. EPA, 1996b).

### **3.4 BEST TECHNOLOGY BY METAL**

The Best Demonstrated Available Technologies, BDATs (See Section 3.3.1) for metals-contaminated RCRA wastes are summarized in Table 10 according to the type of metal

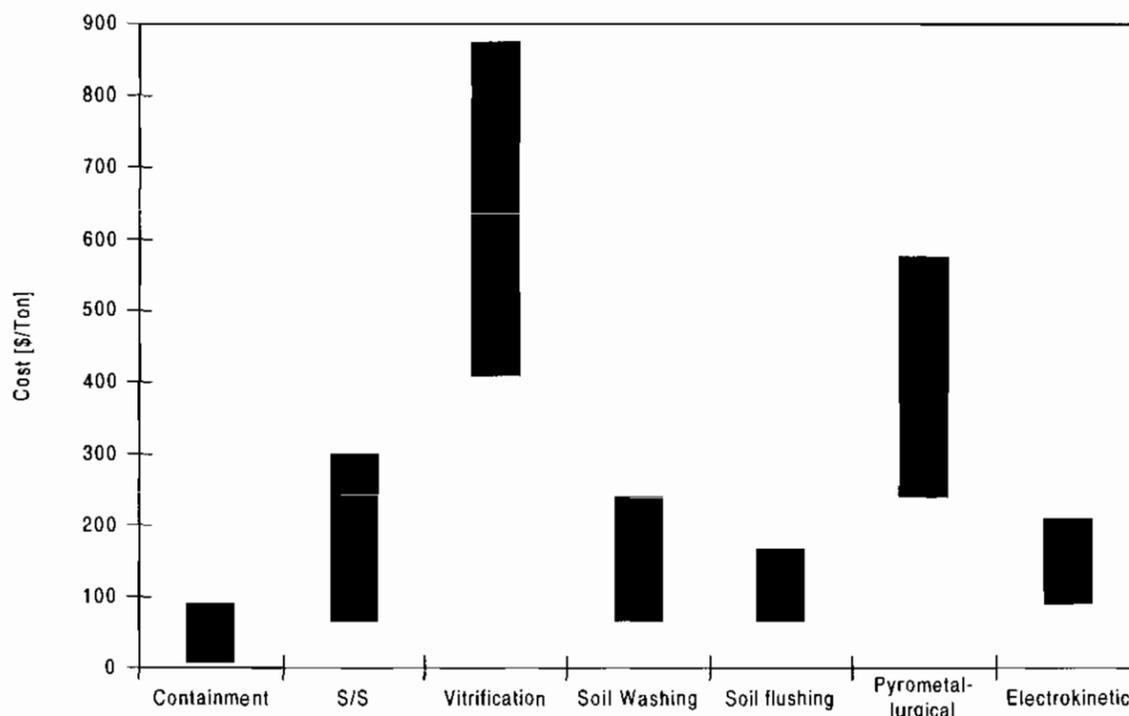
contamination. These technologies can be used as guidelines to review treatment options for Superfund sites, but technology selection at Superfund sites should also consider site-specific characteristics and innovative technologies that may be available under the EPA SITE program.

**Table 10. Summary of Best Demonstrated Available Technologies (BDATs) for RCRA Wastes (from Smith et al., 1995)**

Metal Contaminant	Example BDATs for Metal Wastes	
	Nonwastewater	Wastewater
Lead	Stabilization or metal recovery	Chemical precipitation
Chromium	Chromium reduction and S/S	Chromium reduction and S/S
Arsenic	Vitrification	Chemical precipitation
Cadmium	Stabilization or metal recovery	Chemical precipitation
Mercury	Metal recovery ( $\geq 260$ mg/kg) or acid leaching followed by chemical precipitation	Chemical precipitation with sulfide

## 4.0 COST ESTIMATES

The costs for implementing available technologies will vary significantly between sites because costs are influenced by a wide variety of factors. Figure 4 represents the ranges of operating costs that have been observed for remediation of metals-contaminated soils by a number of techniques that have been discussed. Some important factors influencing costs of specific treatment technologies are discussed below.



**Figure 4. Estimated Operating Costs of Available Remediation Technologies for Metal-Contaminated Soils (U.S. EPA, 1996c)**

### 4.1 CONTAINMENT

The costs associated with capping systems depend largely on the number of components included in the design (Rumer and Ryan, 1995). Barrier and drainage components can add significant amounts to the overall cost of this technology. Sites with steep slopes will also increase cost.

The cost of vertical barrier construction will be influenced by the type of barrier material and the method used to place it. Soil-bentonite trenches provide the most economical method for installation of shallow vertical barriers (Rumer and Ryan, 1995). The most economical deep vertical barrier is a cement-bentonite barrier constructed using a vibrating beam (U.S. EPA 1996b). Costs will also be influenced by ground-water or topographical conditions.

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## 4.2 SOLIDIFICATION/STABILIZATION

Factors directly influencing the costs for implementation of S/S techniques include labor, equipment, energy requirements, testing and monitoring, and the types of reagents. *In situ* processing can lower labor and energy expenses associated with excavation, transport, and disposal of soil from the site.

## 4.3 VITRIFICATION

Treatment costs for *ex situ* vitrification of contaminated soils depend on the waste, throughput capacity of the glass melter, and local energy costs. Site location will affect the cost of transporting the material offsite or equipment transport onsite. As with most technologies, the *in situ* process may provide cost savings over *ex situ* implementation of this technology.

## 4.4 SOIL WASHING

Soil washing at a contaminated site can involve techniques ranging from physical separation and disposal of the contaminated fraction offsite to chemical leaching of contaminants from the entire soil matrix for onsite disposal. Soil washing costs depend largely on the extent to which the contaminated soils are processed.

## 4.5 *IN SITU* SOIL FLUSHING

*In situ* soil flushing involves pumping and treatment of contaminated water, sometimes with recharge of the treated water. The initial and target contaminant concentrations, soil permeability and the depth of the aquifer will influence costs. Chemically enhanced flushing systems will have additional costs associated with reagents and equipment needed to handle the flushing solution. Costs for above-ground treatment of the pumped water vary with contaminant type.

## 4.6 ELECTROKINETIC TREATMENT

The cost of remediating metals-contaminated soils using electrokinetic techniques is strongly influenced by soil conductivity because energy consumption is directly related to the conductivity of the soil between the electrodes. Electrokinetic treatment of soils with high electrical conductivities may not be feasible due to the high cost. Overall expenses for electrokinetic remediation will also be influenced by local energy costs, pretreatment costs, and fixed costs associated with installing the system.

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## 5.0 REGULATORY/POLICY REQUIREMENTS AND ISSUES

Cleanup goals for remediation of metals-contaminated sites vary considerably depending on site-specific factors, especially those that affect the mobility of metals, and regulatory domain. Cleanup goals that are established for a site have a significant influence on determining the acceptability of different technologies for remediation of metals at the site. Thus, the application of remediation technologies to different sites may vary even if the types of contamination at the sites are the same.

A number of states have established soil and ground-water quality criteria that are the basis for cleanup goals. In the absence of such criteria (as in the U.S. Superfund program), or when the criteria are intended as default values, cleanup goals are established based upon site-specific human health and ecological risk assessments which consider the fate and transport of contaminants and possible exposure routes for humans and sensitive environmental receptors. The goals may be established in terms of the total metals in the soil/water or as leachable metals (as defined by various EPA testing procedures). Table 11 provides examples of established cleanup goals for total metals in soils and soil leachate at hazardous waste sites, and Table 12 gives examples of cleanup goals for metals in groundwater.

The use of risk assessment for establishment of site-specific or regional goals for metals in soil or groundwater is difficult because the chemistry of metals is so complex. The hydrogeochemistry of metals is affected by various geochemical and biogeochemical phenomena, including acid-base chemistry, complexation, precipitation/dissolution, adsorption/desorption, and oxidation/reduction. These processes are interlinked and not capable of being described with a simple model. In the case of adsorption/desorption reactions, for example, the speciation of metal ions and the aqueous solution composition determine the extent of reaction (Dzombak and Morel, 1987, 1990). These factors are not captured in a simple partitioning expression. Thus, exposure assessment modeling for metals in soil and groundwater demands the use of flow models integrated with complex chemical models. This requirement frequently has discouraged detailed exposure assessment for metals, resulting in the use of conservative assumptions with regard to metal fate and transport in subsurface systems.

The risk-based corrective action (RBCA) procedure developed by the Environmental Assessment Committee of the American Society for Testing and Materials (ASTM, 1994) may be applied to determine cleanup goals for soil and groundwater. The aim of RBCA is the establishment of cleanup goals based on risk reduction rather than generic cleanup concentrations. However, when regulatory screening levels are exceeded and fate and transport modeling is required as part of a Tier III assessment, there will still be the issue of adequate consideration of the complex chemistry of metals.



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## 6.0 LESSONS LEARNED AND TECHNOLOGY DIRECTIONS

Metals are typically relatively immobile in subsurface systems. For this reason, remediation activities at metals-contaminated sites have focused on the solid-phase sources or repositories of metals. Treatment has often involved excavation of contaminated soil, sludge, or debris followed by *ex situ* treatment or disposal. The most common *ex situ* treatment is solidification/stabilization through addition of chemical reagents, followed by replacement or off-site disposal of the treated material.

Several *in situ* remediation technologies have the potential to provide significant cost savings over *ex situ* techniques because they eliminate the need to excavate and dispose of contaminated solids or to pump and treat contaminated groundwater. *In situ* solidification/stabilization technologies have been demonstrated for treatment of shallow (8-10 ft below surface) wastes and are being implemented at greater depths. Favorable results have been attained using *in situ* vitrification for treatment of a variety of wastes, including metals when metal concentrations do not exceed their glass solubilities. Extraction using *in situ* soil flushing or electrokinetic techniques has been employed at a limited number of sites but may prove to be useful for a range of metal contaminants. Phytoremediation technologies offer promise for remediation of sites with low levels of contamination.

Treatment walls will be used increasingly for effective, low-cost, passive remediation of metal contamination in groundwater. Reactive wall installation will not address metal contamination in soils, but will enable treatment of groundwater contaminated from contact with metal-bearing solids.

Some soil washing technologies are being considered for adaptation to soil leaching/flushing technologies. Chemical additives are being developed to aid with *in situ* extraction of metals from soil.

*In situ* solidification/stabilization techniques are being employed and promise to gain popularity. Application of *in situ* S/S is being aided by development of wide-diameter auger drilling devices that are equipped with chemical reagent delivery systems.

Phytoremediation technologies have only recently gained attention for use in metal remediation. Additional research is needed in order to improve the applicability of phytoremediation for management of metals-contaminated sites. A variety of plants are being investigated for favorable metal accumulation qualities such as a fast rate of uptake.

The future of electrokinetic methods will depend on the efficiency and cost-effectiveness of the technique. Full-scale applications of *in situ* electrokinetic technologies have been initiated in the U.S. but detailed data are not yet available.

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Appendix C

### Analytical Solution Used to Predict the Peak Contaminant Concentration

In the textbook Groundwater, Freeze and Cherry present equations for describing the movement of dissolved contaminants through groundwater aquifers. One of these equations was used to analytically solve for the peak concentration of zinc in groundwater over time at AOC 617 as the plume migrates in the direction of groundwater flow. The equation used (Equation 9.7, page 395) assumes that the contaminant originates as an instantaneous slug at a point source. The mass of contaminant is then carried away from the source by groundwater transport. The assumptions are that the flow field is uniform and steady and that the aquifer is homogeneous and isotropic.

The equation for the peak contaminant concentration that occurs at the center of gravity of the contaminant cloud is:

$$C_{max} = [CoVo]/[8*(\pi*t)^{3/2}*(Dx*Dy*Dz)^{1/2}]$$

Where:

Co= initial concentration (which is assumed to be either the current concentration of 120 mg/L or the maximum observed concentration of 140 mg/L)

Vo= initial volume (which is assumed to be the current limited extent of contaminated groundwater around well F617GW003)

n=porosity (assigned two different values in a sensitivity analysis: 0.3 and 0.4)

t=time

Dx,Dy,Dz=dispersivity values (which are assigned values based upon velocity and typical observed ratios of longitudinal to transverse dispersivity from Freeze & Cherry)

The Excel worksheet illustrates the input parameters used and the shows the output (Cmax) computed. The results indicate that the peak concentration of zinc in groundwater will be less than 5 mg/L (the MCS) after 10 to 20 years of travel time.

<b>Co</b>	<b>140 mg/L</b>	<b>Co</b>	<b>140 mg/L</b>	<b>Co</b>	<b>120 mg/L</b>	<b>Co</b>	<b>140 mg/L</b>
<b>Volume</b>	<b>4246.783 Liters</b>	<b>Volume</b>	<b>4246.783 Liters</b>	<b>Volume</b>	<b>9555.261 Liters</b>	<b>Volume</b>	<b>5662.377 Liters</b>
Vel	0.04 ft/day						
<b>K</b>	<b>2 ft/day</b>						
<b>l</b>	<b>0.02 ft/ft</b>						
<b>n</b>	<b>0.3</b>	<b>n</b>	<b>0.3</b>	<b>n</b>	<b>0.3</b>	<b>n</b>	<b>0.4</b>
<b>alpha L</b>	<b>10</b>	<b>alpha L</b>	<b>30</b>	<b>alpha L</b>	<b>10</b>	<b>alpha L</b>	<b>10</b>
Dx	0.4	Dx	1.2	Dx	0.4	Dx	0.4
<b>alpha T</b>	<b>1</b>						
Dy	0.04	Dy	0.04	Dy	0.04	Dy	0.04
Dz	0.004	Dz	0.004	Dz	0.004	Dz	0.004

<b>Base Case</b>			<b>Change in Dispersivity</b>			<b>Change in Mass</b>			<b>Change in porosity</b>		
Cmax	t (days)	t(years)	Cmax	t (days)	t(years)	Cmax	t (days)	t(years)	Cmax	t (days)	t(years)
239.2459	365	1	138.1287	365	1	461.4029	365	1	318.9946	365	1
84.58621	730	2	48.83587	730	2	163.1305	730	2	112.7816	730	2
46.0429	1095	3	26.58288	1095	3	88.79702	1095	3	61.39053	1095	3
29.90574	1460	4	17.26609	1460	4	57.67536	1460	4	39.87432	1460	4
21.39881	1825	5	12.35461	1825	5	41.26913	1825	5	28.53174	1825	5
16.27862	2190	6	9.398468	2190	6	31.39449	2190	6	21.70483	2190	6
12.91807	2555	7	7.458249	2555	7	24.91341	2555	7	17.22409	2555	7
10.57328	2920	8	6.104484	2920	8	20.39132	2920	8	14.0977	2920	8
8.86096	3285	9	5.115878	3285	9	17.08899	3285	9	11.81461	3285	9
7.56562	3650	10	4.368013	3650	10	14.59084	3650	10	10.08749	3650	10
6.55776	4015	11	3.786125	4015	11	12.64711	4015	11	8.74368	4015	11
5.755362	4380	12	3.32286	4380	12	11.09963	4380	12	7.673817	4380	12
5.104222	4745	13	2.946924	4745	13	9.843856	4745	13	6.805629	4745	13
4.567226	5110	14	2.636889	5110	14	8.808221	5110	14	6.089635	5110	14
4.118202	5475	15	2.377645	5475	15	7.942247	5475	15	5.490936	5475	15
3.738218	5840	16	2.158261	5840	16	7.20942	5840	16	4.98429	5840	16
3.413274	6205	17	1.970655	6205	17	6.582743	6205	17	4.551032	6205	17
3.132823	6570	18	1.808736	6570	18	6.041872	6570	18	4.177097	6570	18
2.888778	6935	19	1.667837	6935	19	5.571214	6935	19	3.851704	6935	19
2.674851	7300	20	1.544326	7300	20	5.158641	7300	20	3.566468	7300	20
2.486083	7665	21	1.435341	7665	21	4.794588	7665	21	3.314777	7665	21
2.318518	8030	22	1.338597	8030	22	4.471428	8030	22	3.091358	8030	22
2.168966	8395	23	1.252253	8395	23	4.183006	8395	23	2.891955	8395	23
2.034828	8760	24	1.174808	8760	24	3.924311	8760	24	2.713104	8760	24
1.913967	9125	25	1.10503	9125	25	3.691223	9125	25	2.551957	9125	25
1.804615	9490	26	1.041895	9490	26	3.480329	9490	26	2.406153	9490	26
1.705293	9855	27	0.984551	9855	27	3.288779	9855	27	2.273723	9855	27
1.614758	10220	28	0.932281	10220	28	3.114177	10220	28	2.153011	10220	28
1.53196	10585	29	0.884478	10585	29	2.954495	10585	29	2.042614	10585	29
1.456004	10950	30	0.840624	10950	30	2.808008	10950	30	1.941339	10950	30

Appendix D

## COMPARISON OF TOTAL COST OF REMEDIAL SOLUTIONS

<b>Site:</b>	Charleston Naval Complex	<b>Base Year:</b>	2002
<b>Location:</b>	AOC 617	<b>Date:</b>	01/11/02
<b>Phase:</b>	Corrective Measures Study		

	Alternative Number 1	Alternative Number 2	Alternative Number 3
<b>Total Project Duration (Years)</b>	30	20	10
<b>Capital Cost</b>	\$12,700	\$298,000	\$235,000
<b>Annual O&amp;M Cost</b>	\$13,000	\$79,000 Yr 1-5 \$19,200 Yr 6-20	\$68,000 Yr 1 \$30,360 Yr 2-10
<b>Total Present Value of Solution</b>	\$256,000	\$790,000	\$410,000

Disclaimer: The information in this cost estimate is based on the best available information regarding the anticipated scope of the remedial alternatives. Changes in the cost elements are likely to occur as a result of new information and data collected during the engineering design of the remedial alternative. This is an order-of-magnitude cost estimate that is expected to be within -50 to +100 percent of the actual project costs.

Alternative: <b>Number 1</b>		<b>COST ESTIMATE SUMMARY</b>				
Elements: <b>Natural Attenuation with Land Use Controls</b>						
<b>Site:</b>	Charleston Naval Complex	<b>Description:</b>	Implementation of base-wide land use management plan to put deed restrictions in place to prevent ingestion of groundwater.			
<b>Location:</b>	AOC 617					
<b>Phase:</b>	Corrective Measures Study					
<b>Base Year:</b>	2002					
<b>Date:</b>	02/01/02					
<b>ANNUAL CAPITAL COSTS</b>						
DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES	
Initial Round of Water Levels	1	EA	\$1,600	\$1,600	See Water Levels Worksheet	
Semi-Annual Groundwater Monitoring of 8 wells	16	EA	\$500	\$8,000	See Laboratory Worksheet	
<b>SUBTOTAL</b>				<b>\$9,600</b>		
Contingency	20%		\$9,600	\$1,920		
<b>SUBTOTAL</b>				<b>\$11,520</b>		
Project Management	10%		\$11,520	\$1,152	USEPA 2000, p. 5-13, <\$100K	
Remedial Design	0%		\$11,520	\$0	Not applicable.	
Construction Management	0%		\$11,520	\$0	Not applicable.	
<b>SUBTOTAL</b>				<b>\$1,152</b>		
<b>TOTAL CAPITAL COST</b>				<b>\$12,700</b>		
<b>OPERATIONS AND MAINTENANCE COST</b>						
DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES	
Annual Groundwater Monitoring of 10 wells	16	EA	\$500	\$8,000		
Reporting	1	EA	\$1,500	\$1,500		
LUC Auditing	1	EA	\$1,000	\$1,000		
<b>SUBTOTAL</b>				<b>\$10,500</b>		
Allowance for Misc. Items	20%		\$10,500	\$2,100		
<b>SUBTOTAL</b>				<b>\$12,600</b>		
<b>TOTAL ANNUAL O&amp;M COST</b>				<b>\$13,000</b>		
<b>PRESENT VALUE ANALYSIS</b>						
			Discount Rate =	3.2%		
End Year	COST TYPE	TOTAL COST	TOTAL COST PER YEAR	DISCOUNT FACTOR (3.2%)	PRESENT VALUE	NOTES
1	FIRST YEAR CAPITAL COST	\$12,700	\$12,700	1.000	\$12,700	
30	ANNUAL CAPITAL COST	\$13,000	\$13,000	18.715	\$243,290	
					\$255,990	
	<b>TOTAL PRESENT VALUE OF ALTERNATIVE</b>				<b>\$256,000</b>	
<b>SOURCE INFORMATION</b>						
1. United States Environmental Protection Agency. July 2000. A Guide to Preparing and Documenting Cost Estimates During the Feasibility Study. EPA 540-R-00-002. (USEPA, 2000).						

Alternative: <b>Number 2</b>		<b>COST ESTIMATE SUMMARY</b>				
Elements: <b>In-Situ Stabilization</b>						
<b>Site:</b>	Charleston Naval Complex	<b>Description:</b>				
<b>Location:</b>	AOC 617	Lime-slurry injection into shallow groundwater zone (5-10 ft bgs);				
<b>Phase:</b>	Corrective Measures Study	effect will be to bring pH into optimal zone for zinc precipitation.				
<b>Base Year:</b>	2002					
<b>Date:</b>	02/01/02					
<b>CAPITAL COSTS</b>						
DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES	
Initial Round of Water Levels	1	EA	\$1,600	\$1,600	See Water Levels Worksheet	
Groundwater monitoring: quarterly of 4 wells for first year	16	EA	\$500	\$8,000	See Laboratory Worksheet	
Pilot study	1	EA	\$20,000	\$20,000	ARS Technologies	
Initial Lime-slurry injection	1	EA	\$200,000	\$200,000	ARS Technologies	
<b>SUBTOTAL</b>				<b>\$229,600</b>		
Contingency	20%		\$229,600	\$45,920		
<b>SUBTOTAL</b>				<b>\$275,520</b>		
Project Management	8%		\$275,520	\$22,042	USEPA 2000, p. 5-13, \$100K \$500K	
Remedial Design	0%		\$275,520	\$0	Included in ARS estimate.	
Construction Management	0%		\$275,520	\$0	Included in ARS estimate.	
<b>SUBTOTAL</b>				<b>\$22,042</b>		
<b>TOTAL CAPITAL COST</b>				<b>\$298,000</b>		
<b>OPERATIONS AND MAINTENANCE COST</b>						
DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES	
Re-injection of additional lime-slurry	1	EA	\$50,000	\$50,000		
GW Monitoring: Qtrly sampling of 8 wells for first five years; semi-annual sampling of 8 wells for subsequent 15 years	32	EA	\$500	\$16,000	See Laboratory Worksheet	
<b>SUBTOTAL</b>				<b>\$66,000</b>		
Allowance for Misc. Items	20%		\$66,000	\$13,200		
<b>SUBTOTAL</b>				<b>\$79,200</b>		
<b>TOTAL ANNUAL O&amp;M COST</b>				<b>\$79,000</b>		
<b>PRESENT VALUE ANALYSIS</b>						
			Discount Rate =	3.2%		
End Year	COST TYPE	TOTAL COST	TOTAL COST PER YEAR	DISCOUNT FACTOR (3.2%)	PRESENT VALUE	NOTES
1	CAPITAL COST	\$298,000	\$298,000	1.000	\$298,000	
5	ANNUAL O&M COST (Yr 1-5)	\$79,000	\$79,000	3.699	\$292,252	
20	ANNUAL O&M COST (Yr 6-20)	\$19,200	\$19,200	10.374	\$199,181	
					\$789,433	
<b>TOTAL PRESENT VALUE OF ALTERNATIVE</b>					<b>\$790,000</b>	
<b>SOURCE INFORMATION</b>						
1. United States Environmental Protection Agency. July 2000. A Guide to Preparing and Documenting Cost Estimates During the Feasibility Study. EPA 540-R-00-002. (USEPA, 2000).						

Alternative: <b>Number 3</b>		<b>COST ESTIMATE SUMMARY</b>				
Elements: <b>Groundwater Extraction with Treatment using Alkaline Precipitation</b>						
<b>Site:</b>	Charleston Naval Complex	<b>Description:</b> Groundwater extraction and treatment with alkaline precipitation. Treated groundwater disposal via sewer.				
<b>Location:</b>	AOC 817					
<b>Phase:</b>	Corrective Measures Study					
<b>Base Year:</b>	2002					
<b>Date:</b>	02/01/02					
<b>CAPITAL COSTS</b>						
DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES	
Initial Round of Water Levels	1	EA	\$1,600	\$1,600	See Water Levels Worksheet	
Laboratory-first year: quarterly influent and effluent samples; monthly discharge analy.	20	EA	\$500	\$10,000	See Laboratory Worksheet	
Recovery Wells	2	EA	\$1,500	\$3,000	Engineers estimate	
Pump Equipment	1	EA	\$2,800	\$2,800	See Pump Worksheet	
Electrical	1	EA	\$10,400	\$10,400	See Electrical Worksheet	
Alkaline Precipitation Treatment	1	EA	\$74,000	\$74,000	See Alkaline Precip. Sheet	
Fence	1	EA	\$5,400	\$5,400	See Site Security Sheet	
Excavation	1	EA	\$40,000	\$40,000		
<b>SUBTOTAL</b>				<b>\$147,200</b>		
Contingency	20%		\$147,200	\$29,440	10% Scope + 10% Bid	
<b>SUBTOTAL</b>				<b>\$176,640</b>		
Project Management	8%		\$176,640	\$14,131	USEPA 2000, p. 5-13, \$100K-\$500K	
Remedial Design	15%		\$176,640	\$26,496	USEPA 2000, p. 5-13, \$100K-\$500K	
Construction Management	10%		\$176,640	\$17,664	USEPA 2000, p. 5-13, \$100K-\$500K	
<b>SUBTOTAL</b>				<b>\$58,291</b>		
<b>TOTAL CAPITAL COST</b>				<b>\$235,000</b>		
<b>OPERATIONS AND MAINTENANCE COST</b>						
DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES	
Laboratory (Qty monitoring 1st year, semi Yr 2-10)	32	EA	\$500	\$16,000		
Monthly discharge analysis	12	EA	\$200	\$2,400		
Pump Equipment	1	EA	\$4,200	\$4,200		
Asphalt excavation, pipe placement, repave	1	EA	\$0	\$0		
Electrical	1	EA	\$3,100	\$3,100		
Alkaline precipitation treatment	1	EA	\$31,080	\$31,080		
Fence	1	EA	\$400	\$400		
<b>SUBTOTAL</b>				<b>\$57,180</b>		
Allowance for Misc. Items	20%		\$57,180	\$11,436		
<b>SUBTOTAL</b>				<b>\$68,616</b>		
<b>TOTAL ANNUAL O&amp;M COST</b>				<b>\$69,000</b>		
<b>PRESENT VALUE ANALYSIS</b>						
			Discount Rate =	3.2%		
End Year	COST TYPE	TOTAL COST	TOTAL COST PER YEAR	DISCOUNT FACTOR (3.2%)	PRESENT VALUE	NOTES
1	CAPITAL COST	\$235,000	\$235,000	1.000	\$235,000	
1	ANNUAL O&M COST-FIRST YR	\$69,000	\$69,000	0.969	\$66,860	
5	ANNUAL O&M COST-Yr 2-5	\$30,360	\$30,360	3.585	\$108,641	
					\$410,701	
<b>TOTAL PRESENT VALUE OF ALTERNATIVE</b>					<b>\$410,000</b>	
<b>SOURCE INFORMATION</b>						
1. United States Environmental Protection Agency. July 2000. A Guide to Preparing and Documenting Cost Estimates During the Feasibility Study. EPA 540-R-00-002. (USEPA, 2000).						

<b>Alternative:</b>	<b>Number 3</b>				
<b>Element:</b>	<b>Pump Installation</b>				
<b>Site:</b>	Charleston Naval Complex	<b>Prepared By:</b>	RLC	<b>Checked By:</b>	
<b>Location:</b>	AOC 617	<b>Date:</b>		<b>Date:</b>	
<b>Phase:</b>	Corrective Measures Study				
<b>Base Year:</b>	2002				
<b>WORK STATEMENT</b>					
Pump groundwater to surface for treatment.					
<b>CAPITAL COSTS</b>					
<b>DESCRIPTION</b>	<b>QTY</b>	<b>UNIT</b>	<b>UNIT COST</b>	<b>TOTAL</b>	<b>NOTES</b>
Pump	1	EA	\$1,550	\$1,550	CH2M-Jones Est.
Install pump and secure	4	HR	\$68	\$272	CH2M-Jones Est.
Controller	1	EA	\$200	\$200	CH2M-Jones Est.
Tubing	20	LF	\$5	\$100	CH2M-Jones Est.
Connections	1	EA	\$50	\$50	CH2M-Jones Est.
Other Consumables	1	EA	\$200	\$200	CH2M-Jones Est.
<b>SUBTOTAL</b>				<b>\$2,372</b>	
Allowance for Misc Items	20%		\$2,372	\$474	
<b>SUBTOTAL</b>				<b>\$2,846</b>	
<b>TOTAL UNIT COST</b>				<b>\$2,800</b>	
<b>OPERATIONS AND MAINTENANCE COST</b>					
<b>DESCRIPTION</b>	<b>QTY</b>	<b>UNIT</b>	<b>UNIT COST</b>	<b>TOTAL</b>	<b>NOTES</b>
Site Operator	52	HR	\$68	\$3,536	1 hr/week for one year
<b>SUBTOTAL</b>				<b>\$3,536</b>	
Allowance for Misc. Items	20%		\$3,536	\$707	
<b>SUBTOTAL</b>				<b>\$4,243</b>	
<b>TOTAL ANNUAL O&amp;M COST</b>				<b>\$4,200</b>	
<b>Source of Cost Data</b>					
1. Sources are as noted in cost table.					

Alternative: **Number 3**  
 Element: **Asphalt Excavation and Pipe Installation**

Site: Charleston Naval Complex  
 Location: AOC 617  
 Phase: Corrective Measures Study  
 Base Year: 2002

Prepared By: RLC  
 Date:

Checked By:  
 Date:

### WORK STATEMENT

Excavate trench through asphalt in order to lay pipe to sanitary sewer line.

### CAPITAL COSTS

DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES
Utility checks and permits	4	EA	\$200	\$800	CH2M-Jones Est.
Asphalt cutting	400	LF	\$1	\$460	CH2M-Jones Est.
Asphalt removal	800	SF	\$3	\$2,520	CH2M Est. - 3-person crew, 8-hrs per day, 40 trees/day
Excavation - machine	120	CY	\$6	\$720	CH2M-Jones Est.
Waste disposal	30	CY	\$260	\$7,800	CH2M-Jones Est.
Clean Fill	30	CY	\$35	\$1,050	CH2M-Jones Est.
Compaction	120	CY	\$12	\$1,420	CH2M-Jones Est.
Replace asphalt	800	SF	\$5	\$4,000	CH2M-Jones Est.
Site Operator-Oversight	80	HR	\$100	\$8,000	CH2M-Jones Est.
4" Sch. 40 PVC	200	LF	\$25	\$5,000	CH2M-Jones Est.
Pipe Connections	10	EA	\$25	\$250	CH2M-Jones Est.
Pipe Handling	20	HR	\$68	\$1,350	CH2M-Jones Est.
<b>SUBTOTAL</b>				<b>\$33,370</b>	
Allowance for Misc. Items	20%		\$33,370	\$6,674	10% Scope + 10% Bid
<b>SUBTOTAL</b>				<b>\$40,044</b>	
<b>TOTAL UNIT COST</b>				<b>\$40,000</b>	

### OPERATIONS AND MAINTENANCE COST

DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES
<b>SUBTOTAL</b>				<b>\$0</b>	
Allowance for Misc. Items	20%		\$0	\$0	
<b>SUBTOTAL</b>				<b>\$0</b>	
<b>TOTAL ANNUAL O&amp;M COST</b>				<b>\$0</b>	

### Source of Cost Data

1. R.S. Means Company. 2000. Environmental Remediation Cost Data - Unit Price, 6th Edition. R.S. Means Company and Talisman Partners, Ltd. Kingston, MA.

Alternative: **Number 3**  
 Element: **Electrical Hookup**

Site: Charleston Naval Complex  
 Location: AOC 617  
 Phase: Corrective Measures Study  
 Base Year: 2002

Prepared By: RLC  
 Date:

Checked By:  
 Date:

### WORK STATEMENT

Install electric power to remediation system of choice.  
 No O&M included here - refer to Land Use Controls

### CAPITAL COSTS

DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES
<b>Electricity Installation</b>					
Install electric panel	1	EA	\$700	\$700	CH2M-Jones Est.
Conduit & Wiring	50	LF	\$70	\$3,500	CH2M-Jones Est.
Electrician	20	HR	\$150	\$3,000	CH2M-Jones Est.
System Connections	10	HR	\$150	\$1,500	CH2M-Jones Est.
<b>SUBTOTAL</b>				<b>\$8,700</b>	
Allowance for Misc. Items	20%		\$8,700	\$1,740	
<b>SUBTOTAL</b>				<b>\$10,440</b>	
<b>TOTAL UNIT COST</b>				<b>\$10,400</b>	

### OPERATIONS AND MAINTENANCE COST

DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES
<b>Annual O&amp;M Costs</b>					
Electricity	20000	kWh	\$0.10	\$2,000	
Electrician	4	HR	\$150	\$600	
<b>SUBTOTAL</b>				<b>\$2,600</b>	
Allowance for Misc Items	20%		\$2,600	\$520	
<b>SUBTOTAL</b>				<b>\$3,120</b>	
<b>TOTAL ANNUAL O&amp;M COST</b>				<b>\$3,100</b>	

### Source of Cost Data

1. R.S. Means Company. 2000. Environmental Remediation Cost Data - Unit Price, 6th Edition. R.S. Means Company and Talisman Partners, Ltd. Kingston, MA.

<b>Alternative:</b>	<b>Numbers 1,2,3</b>				
<b>Element:</b>	<b>Water Levels</b>				
<b>Site:</b>	Charleston Naval Complex	<b>Prepared By:</b>	RLC	<b>Checked By:</b>	
<b>Location:</b>	AOC 617	<b>Date:</b>		<b>Date:</b>	
<b>Phase:</b>	Corrective Measures Study				
<b>Base Year:</b>	2002				
<b>WORK STATEMENT</b>					
Costs associated with a one-time collection of water levels at high and low tides.					
<b>CAPITAL COSTS</b>					
DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES
Collection of water levels at high tide	6	HR	\$40	\$240	CH2M-Jones Est.
Collection of water levels at low tide	6	HR	\$40	\$240	
Potentiometric contour maps	8	HR	\$110	\$880	
<b>SUBTOTAL</b>				\$1,360	
Allowance for Misc. Items	20%		\$1,360	\$272	
<b>SUBTOTAL</b>				\$1,632	
<b>TOTAL UNIT COST</b>				\$1,600	
<b>OPERATION AND MAINTENANCE COSTS</b>					
DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES
<b>SUBTOTAL</b>				\$0	
Allowance for Misc. Items	20%		\$0	\$0	
<b>SUBTOTAL</b>				\$0	
<b>TOTAL O&amp;M COST</b>				\$0	

<b>Alternative: Numbers 1,2,3</b>					
<b>Element: Laboratory Costs</b>					
<b>Site:</b> Charleston Naval Complex	<b>Prepared By:</b> RLC	<b>Checked By:</b>			
<b>Location:</b> AOC 617	<b>Date:</b>	<b>Date:</b>			
<b>Phase:</b> Corrective Measures Study					
<b>Base Year:</b> 2002					
<b>WORK STATEMENT</b>					
Costs associated with water sample collection, shipment and analysis on a per event and per well basis.					
<b>CAPITAL COSTS</b>					
<b>DESCRIPTION</b>	<b>QTY</b>	<b>UNIT</b>	<b>UNIT COST</b>	<b>TOTAL</b>	<b>NOTES</b>
<b>Equipment &amp; Labor per Event</b>					
1 Liter Polypropylene	1	EA	\$1	\$1	CH2M-Jones Est.
Coolers	1	EA	\$10	\$10	CH2M-Jones Est.
Disposable Gloves	1	BOXES	\$20	\$20	CH2M-Jones Est.
Collection of samples	2	HR	\$68	\$136	CH2M-Jones Est.
Sample Shipment	1	EA	\$20	\$20	CH2M-Jones Est.
Sample Analysis (metals)	1	SAMPLE	\$140	\$140	GEL, PEL, STL average
Analysis of data	1	HR	\$100	\$100	CH2M-Jones Est.
<b>SUBTOTAL</b>				<u>\$427</u>	
Allowance for Misc. Items	20%		\$427	\$85.40	
<b>SUBTOTAL</b>				<u>\$512</u>	
<b>TOTAL UNIT COST</b>				<b>\$500</b>	
<b>OPERATION AND MAINTENANCE COSTS</b>					
<b>DESCRIPTION</b>	<b>QTY</b>	<b>UNIT</b>	<b>UNIT COST</b>	<b>TOTAL</b>	<b>NOTES</b>
<b>SUBTOTAL</b>				\$0	
Allowance for Misc. Items	20%		\$0	\$0	
<b>SUBTOTAL</b>				<u>\$0</u>	
<b>TOTAL O&amp;M COST</b>				<b>\$0</b>	
<b>Source of Cost Data</b>					
1. Analytical Bid Form - Charleston Naval Complex - Level II					

<b>Alternative:</b>	<b>Number 3</b>					
<b>Element:</b>	<b>Fencing</b>					
<b>Site:</b>	Charleston Naval Complex	<b>Prepared By:</b> RLC	<b>Checked By:</b>			
<b>Location:</b>	AOC 617	<b>Date:</b>	<b>Date:</b>			
<b>Phase:</b>	Corrective Measures Study					
<b>Base Year:</b>	2002					
<b>WORK STATEMENT</b>						
Implement land use controls, including fencing to restrict access and signs. O&M includes periodic inspection.						
<b>CAPITAL COSTS</b>						
	<b>DESCRIPTION</b>	<b>QTY</b>	<b>UNIT</b>	<b>UNIT COST</b>	<b>TOTAL</b>	<b>NOTES</b>
	<b>Fence</b>					
	Install Fence	150	LF	\$10.00	\$1,500	CH2M-Jones Est.
	sandbags	100	EA	\$4.00	\$400	CH2M-Jones Est.
	<b>SUBTOTAL</b>				<u>\$1,900</u>	
	General requirements	7%		\$1,900.00	\$133	
	Allowance for Misc. Items	20%		\$1,900	<u>\$380</u>	
	<b>SUBTOTAL</b>				<u>\$2,413</u>	
	<b>Signs</b>					
	Install signs	5	EA	\$500.00	\$2,500	CH2M-Jones Est.
	<b>SUBTOTAL</b>				<u>\$2,500</u>	
	Allowance for Misc. Items	20%		\$2,500	<u>\$500</u>	
	<b>SUBTOTAL</b>				<u>\$3,000</u>	
	<b>TOTAL UNIT COST</b>				<u>\$5,400</u>	
<b>OPERATIONS AND MAINTENANCE COST</b>						
	<b>DESCRIPTION</b>	<b>QTY</b>	<b>UNIT</b>	<b>UNIT COST</b>	<b>TOTAL</b>	<b>NOTES</b>
	Fence and sign-Inspections	4	HR	\$30	\$120	CH2M Est. 4 hrs/year
	Misc. Repairs/Maintenance	8	HRS	\$30	\$240	CH2M Est. - 8 hrs/yr
	<b>SUBTOTAL</b>				<u>\$360</u>	
	Allowance for Misc Items	20%		\$360	<u>\$72</u>	
	<b>SUBTOTAL</b>				<u>\$432</u>	
	<b>TOTAL ANNUAL O&amp;M COST</b>				<u>\$400</u>	
<b>Source of Cost Data</b>						
1. R.S. Means Company. 2000. Environmental Remediation Cost Data - Unit Price, 6th Edition. R.S. Means Company and Talsman Partners, Ltd. Kingston, MA.						

**Alternative:** Number 1,3  
**Element:** Alkaline Precipitation System

**Site:** Charleston Naval Complex  
**Location:** AOC 617  
**Phase:** Corrective Measures Study  
**Base Year:** 2002

**Prepared By:** RLC  
**Date:**

**Checked By:**  
**Date:**

### WORK STATEMENT

Use an alkaline precipitation system for remediation of zinc in groundwater

### CAPITAL COSTS

DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES
<b>Capital Costs</b>					
System: Tanks Feed system Mixer Associated Piping	1	EA	\$50,000	\$50,000	McGrane/CH2M HILL
Piping: Tank to Treatment	10	FT	\$100	\$1,000	CH2M JonesEst.
Piping: Treatment to Second Tar	10	FT	\$100	\$1,000	CH2M JonesEst.
Filter Press	1	EA	\$10,000	\$10,000	CH2M JonesEst.
<b>SUBTOTAL</b>				\$62,000	
Allowance for Misc Items	20%		\$62,000	\$12,400	
<b>SUBTOTAL</b>				\$74,400	
<b>TOTAL COST</b>				<b>\$74,000</b>	

### OPERATION AND MAINTENANCE COSTS

DESCRIPTION	QTY	UNIT	UNIT COST	TOTAL	NOTES
Site Operator	300	HR	\$68	\$20,400	CH2M Jones Est. 12 hrs every week for 6 mos
Chemicals	50	DRUM	\$100	\$5,000	CH2M JonesEst.
Non-Hazardous Precipitate Disposal	1	TON	\$500	\$500	CH2M JonesEst.
<b>SUBTOTAL</b>				\$25,900	
Allowance for Misc Items	20%		\$25,900	\$5,180	
<b>SUBTOTAL</b>				\$31,080	
<b>TOTAL ANNUAL O&amp;M COST</b>				<b>\$31,000</b>	

### Source of Cost Data

1. R.S. Means Company. 2000. Environmental Remediation Cost Data - Unit Price, 6th Edition. R.S. Means Company and Talisman Partners, Ltd. Kingston, MA. (Means(a)).
2. R.S. Means Company. 1999. Site Work and Cost Data, 18th Edition. R.S. Means Company. Kingston, MA. (Means(b)).
3. R.S. Means Company. 1999. Heavy Construction Cost Data, 13th Edition. R.S. Means Company. Kingston, MA. (Means(c)).

### Cost Adjustment Checklist

FACTOR:	NOTES:
<input checked="" type="checkbox"/> H&S Productivity	Assume work conducted in Level D
<input checked="" type="checkbox"/> Escalation to Base Year	Current year (2001) is base year
<input checked="" type="checkbox"/> Area Cost Factor	Adjusted Unit Costs for Charleston, South Carolina where applicable
<input checked="" type="checkbox"/> Subcontractor Overhead and Profit	Assumed included in unit prices (15% Overhead + 10% Profit)
<input checked="" type="checkbox"/> Prime Contractor Overhead and Profit	Included in <u>Solution Set</u> Cost Estimates only.