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US Army Corps of Engineers
Waterways Experiment Station

Options for Treatment and Disposal of Contaminated Sediments from New York/ New Jersey Harbor

by Tetra Tech, Inc.

Daniel E. Averett
Environmental Laboratory

Prepared for U.S. Army Engineer District, New York

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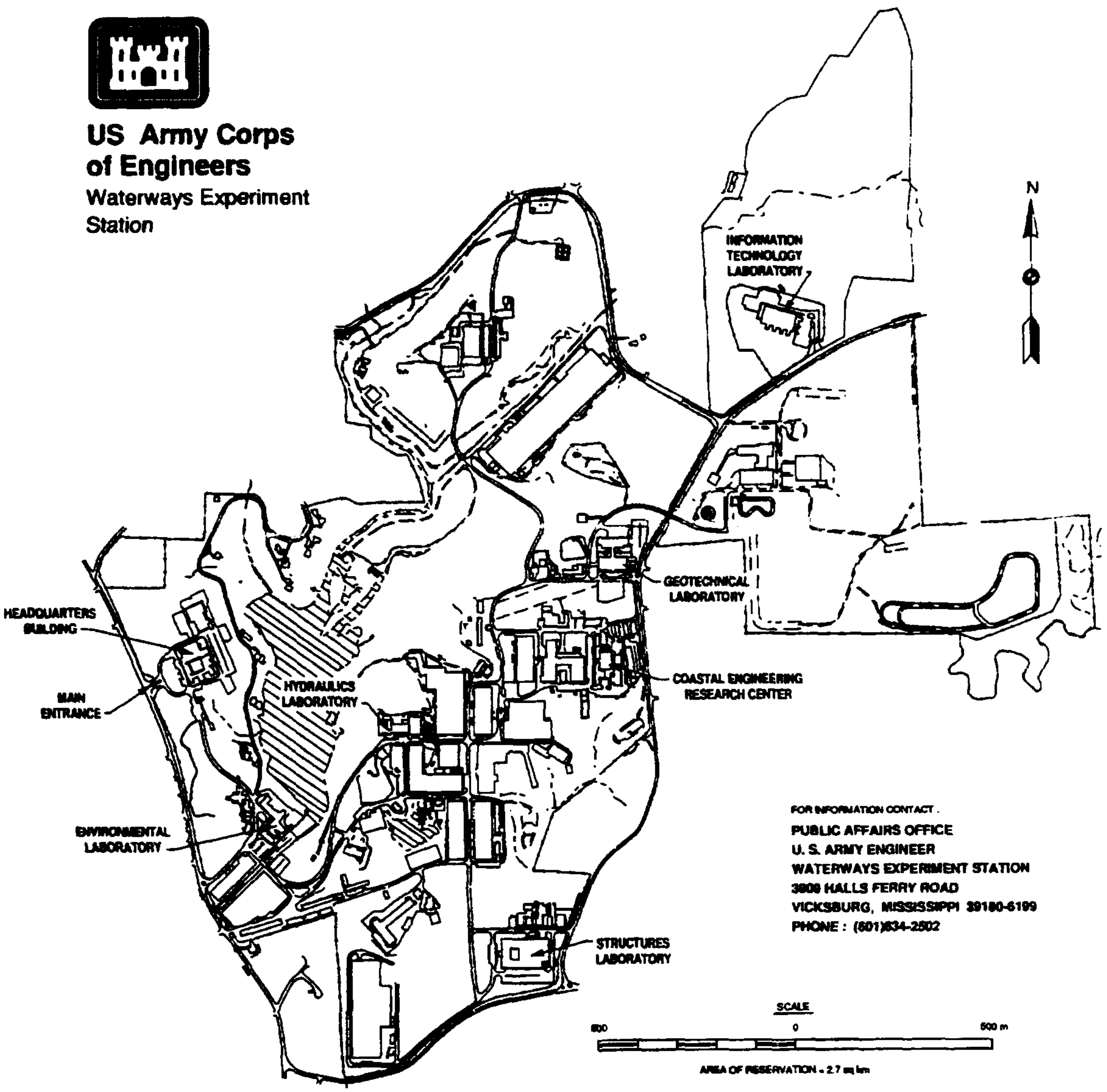

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Preface

The work reported herein was conducted for the U.S. Army Engineer District, New York, by the Environmental Laboratory (EL), U.S. Army Engineer Waterways Experiment Station (WES), and by Tetra Tech, Inc., and EA Engineering, Science, and Technology, Inc., as part of a contract between Tetra Tech and WES (Contract No. DACW39-92-D-0005 for engineering studies in support of dredging and dredged material management).

Funding for this review, assessment, and bench-scale demonstration of selected treatment and disposal options for contaminated sediment from New York/New Jersey Harbor was provided through Section 412(c) of the Water Resources Development Act of 1990. Additional bench-scale and pilot-scale studies of decontamination technologies for harbor sediments are being planned under Section 405 of the Water Resources Development Act of 1992.

The report was written by scientists and engineers at Tetra Tech, Inc., and EA Engineering, Science, and Technology, Inc. Mr. Thomas Grieb was Tetra Tech's project leader; Tetra Tech's lead author and editor was Mr. Kim A. Brown. Mr. Daniel E. Averett, Environmental Engineer, EL, was the technical monitor for the Tetra Tech contract and was a contributing author to the report. Mr. Joe Olha, Oceanographer, Operations Division, was the technical monitor for the New York District.

The work was performed under the general supervision of Mr. Norman R. Francingues, Jr., Chief, Environmental Restoration Branch, Environmental Engineering Division (EED), EL. The Chief of EED was Dr. Raymond L. Montgomery, and the Director of EL was Dr. John Harrison.

At the time of publication of this report, Director of WES was Dr. Robert W. Whalin. Commander was COL Bruce K. Howard, EN.

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Conversion Factors, Non-SI to SI Units of Measurement

Non-SI units of measurement used in this report can be converted to SI units as follows:

Multiply	By	To Obtain
acres	4,046.873	square meters
cubic yards	0.7645549	cubic meters
feet	0.3048	meters
gallons (U.S. liquid)	3.785412	liters
inches	2.54	centimeters
knots (international)	0.5144444	meters per second
miles (U.S. statute)	1.609347	kilometers
tons (2,000 pounds, mass)	907.1847	kilograms

1 Introduction

Background

The U.S. Army Engineer District, New York, dredges more than 5 million yd^3 of sediment each year on average to maintain harbors and waterways for New York and New Jersey Harbor.[1] In the past, most of this dredged material had characteristics that allowed its disposal into open water or ocean sites. However, more accurate detection of toxic contaminants in estuarine sediments and the advocation of more restrictive criteria for ocean disposal have led to the recent investigation of other dredged material management alternatives. Alternatives for material that is unacceptable for open-ocean disposal include ocean disposal with capping, coastal borrow pit disposal with capping, land-based or in-water confined disposal, and treatment of sediment to reduce the contaminant concentrations to levels acceptable for unrestricted disposal or beneficial uses.

The primary contaminants of concern for this study are tetrachlorodibenzodioxins (TCDD). These chemicals, along with polychlorinated biphenyls (PCBs), have been detected in New York/New Jersey Harbor sediments, and their presence in the tissue of fish and shellfish has resulted in fishing advisories in the waters of the estuary (Squibb, O'Connor, and Kneip 1991). Ocean disposal of dioxin-containing dredged material is controlled by regulation. Dredged sediments that are unacceptable for any type of ocean disposal must be disposed at suitable containment island or on-land sites or be treated to reduce the contaminant concentration to below ocean disposal regulatory levels.

The preliminary assessment of existing treatment technologies and disposal alternatives for dioxin-contaminated sediment from New York/New Jersey Harbor is the primary purpose of this report. Included in the assessment of treatment technologies are a survey of available options, bench-scale tests of selected treatment technologies, development of the overall process train for promising treatment alternatives, an assessment of the feasibility of implementing the alternatives, preliminary cost estimates,

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[1] A table of factors for converting Non-SI units of measurement to SI units is presented on page viii.

and comparison of alternatives. Disposal alternatives have been considered by others (Conner et al. 1979; U.S. Army Corps of Engineers (USACE) New York District 1988; New York University 1989; Walski and Schaefer 1988). Results of these evaluations of disposal options are included in this report to provide a basis for comparison to the treatment alternatives.

Report Organization

This chapter describes the report's purpose and contents, summarizes dredging operations in New York/New Jersey Harbor, discusses the characteristics of dioxins and the environmental concerns associated with dioxins, and introduces the contaminated sediment management options considered in the report.

The second chapter presents a survey of available remediation treatment technologies for dioxin-contaminated New York/New Jersey Harbor sediments. This survey is based on the extensive literature that has been developed in recent years as a result of contaminant remediation investigations and demonstrations throughout the country (Averett et al. 1990; U.S. Congress 1991; and U.S. Environmental Protection Agency (USEPA) Superfund Innovative Technology Evaluation (SITE) programs). General information is provided for six types of treatment technologies with tested or potential application for the remediation of dioxin-contaminated sediments. This information includes descriptions of one or more process options for each technology type in terms of effectiveness, pretreatment and posttreatment requirements, availability, and processing rates.

Based on the technology review, six treatment technologies were selected for more detailed evaluation in Chapter 3. Chapter 4 presents and discusses the three disposal options considered in comparison with the treatment options. Bench-scale studies on dioxin-contaminated New York/New Jersey Harbor sediments were performed for four of the treatment technologies. Results of the treatability studies are incorporated into the evaluation of treatment alternatives in Chapter 5. Both treatment and disposal options are compared, and the technical feasibility of full-scale application of these alternatives to New York/New Jersey Harbor sediments is discussed. The chapter also presents recommendations for future investigations based on the conclusions derived from the bench-scale treatability test results.

Summary of Dredging Operations in NY/NJ Harbor

Dredging

New York/New Jersey Harbor handles the largest volume of cargo of any port in the United States. In 1991, the Harbor processed nearly 12 million tons of container cargo valued at $4.6 billion.[1] The New York University estimated the regional economic impact of port operations at $19.6 billion, providing $5.2 billion in direct wages and creating 181,000 jobs. Harbor facilities include 750 miles of waterfront, 2,600 acres of marina facilities, and over one million linear feet of berthage (New York University 1989). Forty-five Federal navigation projects totaling 240 miles of channels provide access to these facilities.

The bay areas surrounding New York/New Jersey Harbor are naturally shallow, acting as catchments for river-transported sediments and solids from surface point and nonpoint sources. Without dredging, the main channel depth in New York/New Jersey Harbor would likely be less than 20 ft. Dredging is required to maintain navigability for large cargo ships that require water depths up to 45 ft for operation (New York University 1989).

Approximately 75 percent of the dredging in New York/New Jersey Harbor is conducted by the U.S. Army Corps of Engineers (USACE), with private marinas, industrial berthing facilities, and the Port Authority accounting for most of the remainder. The annual average volume of material dredged from the Harbor between 1970 and 1991 was over 8 million yd^3, ranging from 19.5 million yd^3 in 1971 to 2.3 million yd^3 in 1981 (New York University 1989). Since 1989, when a total of more than 15 million yd^3 w dredged, annual totals have been less than 6 million yd^3.[2]

Disposal operations

Virtually all of the materials dredged from New York/New Jersey Harbor are disposed of at an area 6 n.m. (11 km) east of Sandy Hook, NJ, and 12 n.m. (22 km) south of Rockaways, NY.[3] This area, known as the Mud Dump Site (Figure 1), has been the primary dredged material disposal site in the area since the early 1900s. As a result, sediments in some of the

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[1] Personal Communication, December 1992, K. Durka, Port Authority of New York and New Jersey.

[2] Personal Communication, 7 December 1992, B. May, USACE, New York District, New York, NY.

[3] Personal Communication, 4 December 1992, M. Greges, USACE, New York District, New York, NY.

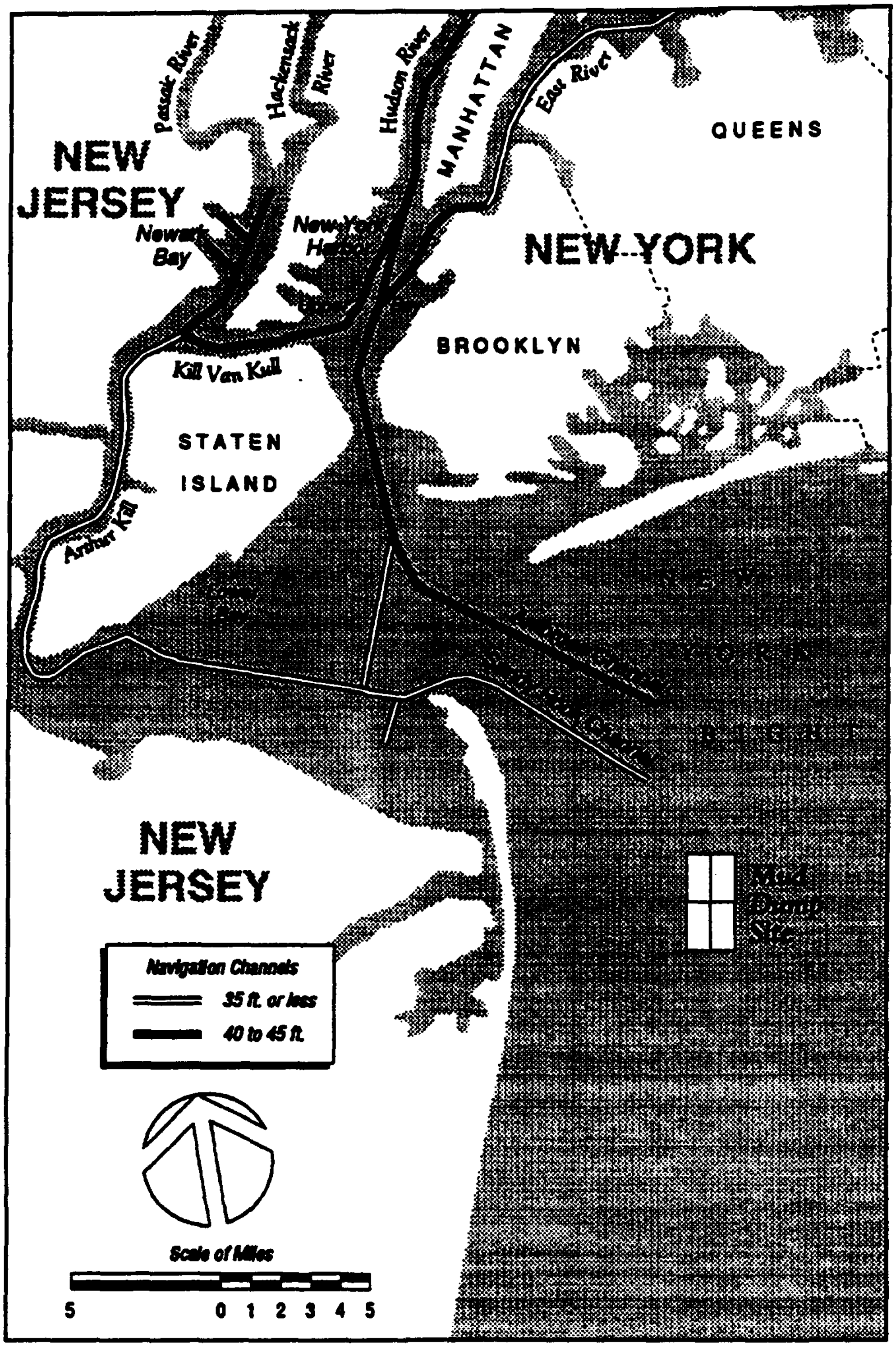

Figure 1. New York/New Jersey Harbor and Mud Dump Site

mounds at the site have accumulated to a height of 30 ft (9.1 m) above the ocean floor.

Sediments with acceptable toxicity or bioaccumulation characteristics are suitable for unrestricted ocean disposal at the Mud Dump Site. Historically, these sediments comprise 90 percent or more of the total material dredged from the Harbor. Unrestricted ocean disposal allows the materials to be deposited at the designated site without any treatment or containment requirements. Transport to the disposal site is generally accomplished by a hopper dredge or by a split-hull barge.[1]

Sediments that do not meet the criteria for unrestricted disposal, but do not represent unacceptable mortality to organisms, may be disposed of at the Mud Dump Site if the materials are capped with approximately 1 m of clean sediments after disposal. These materials have, in the past, represented less than 10 percent of dredged material in any given year. Because the Mud Dump Site is approaching its authorized capacity, it will likely reach closure at some point during the next 5 years, requiring designation of an alternate disposal site.[2]

For more contaminated materials that may be deemed unsuitable for ocean disposal with or without capping, alternate disposal and/or treatment technologies are potential options. This report reviews these options and provides a preliminary assessment of their technical feasibility.

Sediment characteristics

Sediment particle size varies considerably throughout the channels and harbors; but in general, sediments in the higher energy areas, such as the major approach channels, are predominantly sand, while sediments from the bays and berthage areas are predominantly silts and clays. A study of sediments from Federal dredging projects in 1977 found these sediments to be silt-clay material (67 percent) and sand (33 percent). A number of contaminants (e.g., cadmium, copper, zinc, lead, mercury, PCBs, polyaromatic hydrocarbons (PAHs), chlorinated pesticides, and other halogenated hydrocarbons) have been detected in sediments collected in the New York/New Jersey Harbor area (Squibb, O'Connor, and Kneip 1991). These contaminants enter the system from a variety of sources including municipal and industrial discharges, urban runoff, landfill sites, and accidental spills (New York University 1989). Because of the greater affinity of contaminants for small particles, fine-grained sediments tend to have higher sediment contaminant concentrations than sand.

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[1] Personal Communication, 5 February 1993, M. Greges, USACE, New York District, New York, NY.

[2] Personal Communication, 7 December 1992, B. May, USACE, New York District, New York, NY.

The detection of dioxins in New York/New Jersey Harbor sediments has generated considerable concern recently because of their potential for adverse biological effects. Analyses of dioxin concentrations in the Harbor sediments are somewhat limited (Squibb, O'Connor, and Kneip 1991). Much of the work done to date has focused on concentrations in the Newark Bay Complex, which includes Newark Bay, Arthur Kill, and the Passaic River, areas near a former chemical manufacturing plant located in Newark, NJ. These studies do not provide sufficient data to characterize the distribution of dioxins throughout the harbor, but the concentrations observed in these studies have generated concern.

A review of results of sediment dioxin studies by Bopp et al. (1991), Pruell et al. (1990), and Finley et al. (1990) provides an indication of the dioxin levels that may be considered appropriate for application of treatment or more restrictive disposal technologies. Reported dioxin concentrations have tended to be lower in near-surface samples (0 to 4 cm) compared with deeper sediments, primarily for areas where dredging for navigational purposes has not been routinely performed. Concentrations in samples collected at 10 locations in the Passaic River, Newark Bay, Kill van Kull, and New York Harbor ranged from 20 parts per trillion (pptr)[1] to 2,700 pptr in the top 4 cm of sediment (Bopp et al. 1991). The highest concentrations reported, 21,000 pptr, were collected in the Passaic River at a sediment depth of 108 to 111 cm (Bopp et al. 1991). Pruell et al. (1990) reported that near-surface dioxin concentrations from the lower Passaic River ranged from 482 to 759 pptr. Dioxin concentrations measured in 19 surface samples collected in 1990 from the lower Passaic River and from Newark Bay ranged from below detection limit (1.6 pptr) to 510 pptr (Finley et al. (1990) as cited in Wenning et al. (1992)).

Environmental Concerns of Dioxins

Dioxins are examples of chlorinated organic compounds, which are persistent in the environment, can bioaccumulate, and can cause toxic effects in humans and biota. Sources of dioxins include impurities formed during manufacture of other organic compounds (e.g., 2,4,5 trichlorophenoxy herbicides, hexachlorophene, pentachlorophenol, and PCBs); by-products of chlorine bleaching processes used in some paper and pulp mills; incineration of municipal and industrial wastes; fires; other industrial processes involving use of chlorine or chlorinated solvents; and automobile exhaust from leaded gasoline (Agency for Toxic Substances and Disease Registry (ATSDR) 1987; Palmer et al. 1988). Polychlorinated dibenzo–p– dioxins (PCDDs) occur as 75 congeners with different numbers of chlorine atoms in different positions. Table 1 lists the most common isomers. The most

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[1] Sediment contaminant concentrations in this report are presented on a dry weight basis, i.e., mass of contaminant per mass of dry solids in the sediment. Conversion of the dry weight contaminant concentration to a wet weight basis requires multiplying the dry weight concentration by the dry solids fraction of the wet (in situ) sediment.

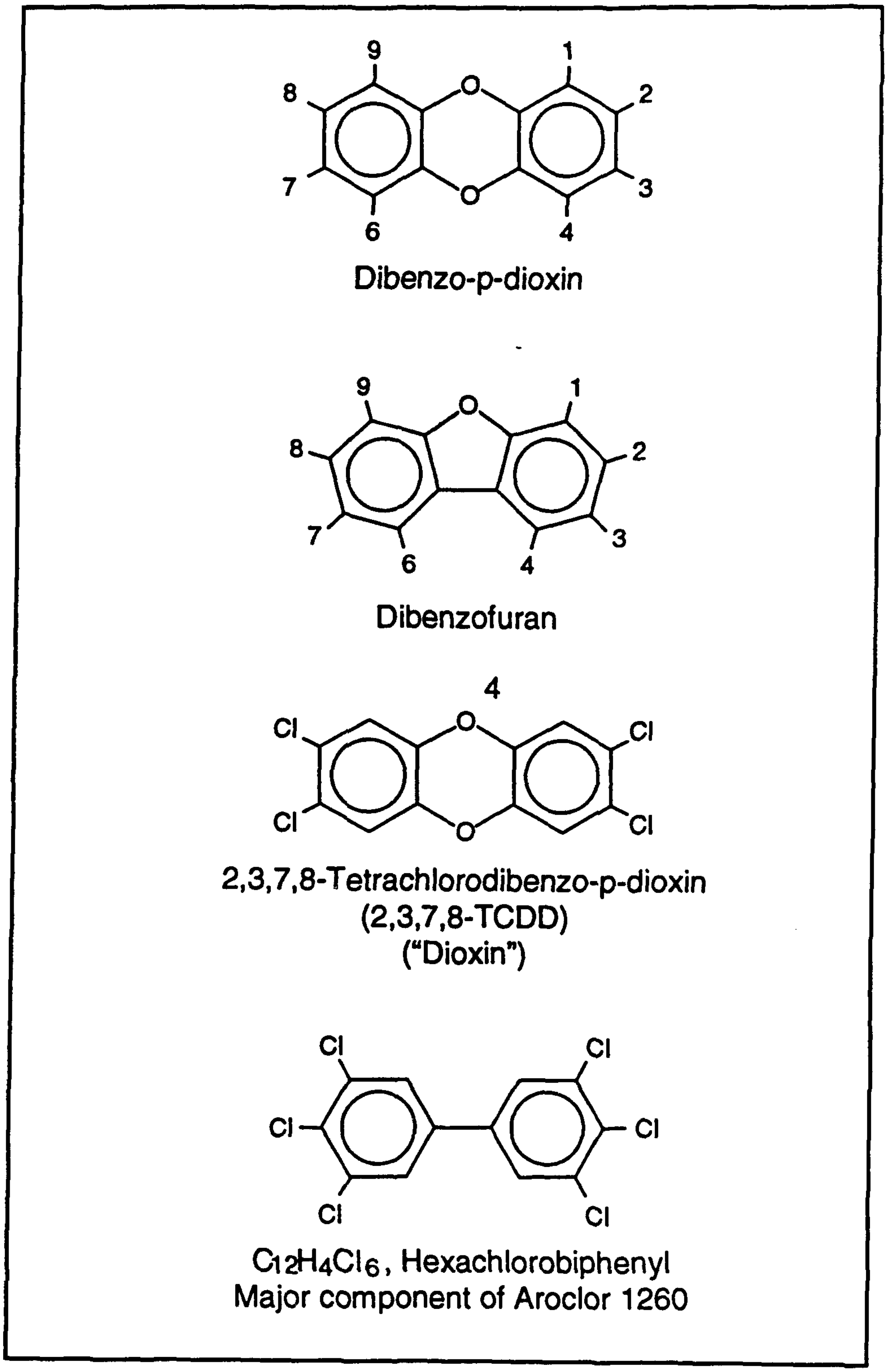
toxic and most studied isomer is 2,3,7,8 tetrachlorodibenzo-p-dioxin (Figure 2). The chemical properties of the congeners and their toxicity depend on the number of chlorine atoms present and where the chlorine atoms are located, as discussed below.

Table 1
CDD and CDF Isomer Groups, Isomers, and Congeners

CDDs	Number of Isomers in Group
1. Monochlorodibenzodioxin (monoCDD)	2
2. Dichlorodibenzodioxin (diCDD)	10
3. Trichlorodibenzodioxin (triCDD)	14
4. Tetrachlorodibenzodioxin (tetraCDD)	22
5. Pentachlorodibenzodioxin (pentaCDD)	14
6. Hexachlorodibenzodioxin (hexaCDD)	10
7. Heptachlorodibenzodioxin (heptaCDD)	2
8. Octachlorodibenzodioxin (octaCDD)	1
Total CDD Congeners	75
CDFs	
1. Monochlorodibenzofuran (monoCDF)	4
2. Dichlorodibenzofuran (diCDF)	16
3. Trichlorodibenzofuran (triCDF)	28
4. Tetrachlorodibenzofuran (tetraCDF)	38
5. Pentachlorodibenzofuran (pentaCDF)	28
6. Hexachlorodibenzofuran (hexaCDF)	16
7. Heptachlorodibenzofuran (heptaCDF)	4
8. Octachlorodibenzofuran (octaCDF)	1
Total CDF Congeners	135

Dioxins not only occur as mixtures of congeners but are often found along with other chlorinated compounds such as furans that have related chemical structures (see Figure 2). Polychlorinated dibenzofurans (PCDFs) consist of 135 congeners with similar characteristics to the PCDDs. PCDFs are also generated as impurities in other organic compounds (e.g., PCBs, hexachlorobenzene, pentachlorophenol, and phenoxy herbicides) and can be generated during incineration at a temperature less than 800 °C (USEPA 1986a). Less information is available on properties and toxicities of the furans than the dioxins.

In general, dioxins and furans have low solubilities in water, adsorb strongly to organic matter, and have high potential to bioaccumulate. Because of these characteristics, these compounds tend to be concentrated in sediments and lipids (fatty tissue). Specific properties vary within a given group depending on the number of chlorines present and their position. Octanol-water partition coefficients are inversely proportional to water solubilities. These compounds are also highly soluble in organic solvents. Chemical properties of the dioxins are listed in Table 2. Environmental processes that can degrade dioxins include photolysis in surface water or the top few inches of soil (USEPA 1985a,b) and biodegradation, but at a



Dibenzo-p-dioxin

Dibenzofuran

2,3,7,8-Tetrachlorodibenzo-p-dioxin
(2,3,7,8-TCDD)
("Dioxin")

$C_{12}H_4Cl_6$, Hexachlorobiphenyl
Major component of Aroclor 1260

Figure 2. **Chemical structures of selected compounds**

slow rate (ATSDR 1987). PCDDs and PCDFs can be transported across the soil or water interface to air (Palmer et al. 1988).

Table 2
Physical/Chemical Properties of 2,3,7,8 TCDD

	Value	Reference
Molecular weight, g/mol	321.97	Windholtz 1983
Physical state @ 20 °C	Solid, colorless needles with no odor	Windholtz 1983
Melting point, °C	305	Schroy, Hileman, and Cheng 1985
Boiling point, °C	412.2	Schroy, Hileman, and Cheng 1985
Density, g/ml	1.827 (est.)	Schroy, Hileman, and Cheng 1985
Water solubility, S, mg/L	1.93×10^{-5} (22 °C) 3.17×10^{-4} (25 °C)	Marple, Brunck, and Throop 1986 Schroy, Hileman, and Cheng 1985
Vapor pressure, P, mm Hg	1.4×10^{-9} (25 °C)	Schroy, Hileman, and Cheng 1985
Henry's Law Constant, H @ 25 °C (atm • m^3/mol)	2.1×10^{-6} (in water)	Schroy, Hileman, and Cheng 1985
Log (octanol-water partition coefficient), log K_{ow}	6.15 - 7.28	USEPA 1985a
Soil adsorption coefficient, log K_{oc}, ml/g	6.0 - 7.39	USEPA 1985a
Fish bioconcentration factor, BCF	Best estimate, 5000[1] 7900-9300, fathead minnows	USEPA 1985a USEPA 1985b

[1] Value used for estimating aquatic effects

The toxicities of these groups of organic compounds differ among the various congeners. The most toxic of the PCDDs and PCDFs consist of six PCDDs and nine PCDFs. Relative toxicities among these compounds differ as illustrated in Table 3, which shows the 1989 toxicity equivalency factors (TEF). The combined effect of PCDDs and PCDFs is estimated by multiplying the concentration of each congener by its TEF and summing all the resulting values for the 15 compounds. This combined number is referred to as toxicity-equivalent concentration or toxicity-equivalent quotient. This procedure was developed by an international forum that included the USEPA and was intended as an interim method to determine the validity of the present slope factor used to estimate carcinogenic risks and the reference dose used to estimate the potential for systemic toxic effects.

Table 3
1989 Toxicity Equivalency Factors

Compound	TEFs/89
Mono-, Di-, and Tri-CDDs	0
2,3,7,8 TCDD (Other TCDDs)	1 (0)
2,3,7,8 PeCDD (Other PeCDDs)	0.5 (0)
2,3,7,8 HxCDD (Other HxCDDs)	0.1 (0)
2,3,7,8 HpCDD (Other HpCDDs)	0.01 (0)
OCDD	0.001
Mono-, Di-, and Tri-CDFs	0
2,3,7,8 TCDF (Other TCDFs)	0.1 (0)
1,2,3,7,8 PeCDF 2,3,4,7,8 PeCDF (Other PeCDFs)	0.05 0.5 (0)
2,3,7,8 HxCDF (Other HxCDFs)	0.1 (0)
2,3,7,8 HpCDF (Other HpCDFs)	0.01 (0)
OCDF	0.001

The chemical characteristics of dioxins and furans may influence the selection of treatment options for the remediation of contaminated sediments. For example, because they are hydrophobic, they accumulate in sediment, especially in organic matter and clays. Detailed chemical analyses of the sediments are needed to quantify the specific congeners present. Determining the feasibility of chemical treatment methods (e.g., dechlorination reactions or photolysis) depends on the relative amounts of the low and highly chlorinated compounds.

Overview of Management Options for Contaminated Sediment

Arrangement options for contaminated sediments are dependent upon the type of contamination and the concentrations of the contaminants. This study considers only dioxin contamination. Four conceptual options can be defined as follows:

- Leave the contaminated sediment in place.

- Dredge and dispose the sediment at restricted or unrestricted open-water sites (e.g., Mud Dump Site or borrow-pits).

- Dredge and dispose the sediment at a confined disposal facility on land or a confined disposal island.

- Dredge the sediment and treat it to remove or destroy the contaminants to such an extent that it becomes suitable for disposal or beneficial use.

This report addresses the last three options, discussing disposal alternatives and a comparison to a more detailed discussion of selected treatment options. At present, no full-scale treatment option has been demonstrated to be effective with dioxin-contaminated sediments. This study presents what are presently the most promising treatment alternatives, some of which may become viable management options in the future.

2 Treatment Technologies Survey

Treatment technologies with tested or potential application for the remediation of dioxin-contaminated sediments include the following:

- Biological treatments.
- Chemical treatments.
- Extraction processes.
- Immobilization processes.
- Radiant energy treatments.
- Thermal treatments.

A summary of these technologies, based on the classification system of Averett et al. (1990), is presented in Table 4.

Although dioxin-contaminated sediments from New York/New Jersey Harbor are the focus of this study, technologies that have been tested with PCBs alone have also been considered; they are organic contaminants most similar to dioxins and furans, and PCB-contamination of soils and sediments has been subject to considerably more testing than dioxins.

The relationship of treatment options in the overall effort of remediating sediments is shown in Figure 3 where the chosen treatment technology is one of many components in the treatment train.

The information in Table 4 was obtained from various sources: recent publications from the SITE Program of the USEPA Risk Reduction Engineering Laboratory (RREL) (USEPA 1991a); from USACE (Averett et al. 1990); from the Office of Technology Assessment of the U.S. Congress (U.S. Congress 1991); recent literature; from discussions with technology developers and vendors; and from USEPA SITE project managers.

Table 4
Treatment Alternatives for Remediation of Contaminated Sediments

Process	Previous Testing of			Efficiency %	Pretreatment Requirements	Posttreatment Requirements	Availability	Estimated Costs $\$/yd^3$	Process Rates yd^3/day
	Sediments	Dioxins	PCBs						
Biological Treatments									
Bioremediation	Bench	Bench	Field	<99	Mixing	Effluent	Available	N/A	N/A
Chemical Treatments									
Dechlorination									
APEG-PLUS		Bench	Commercial	>99	Drying	None	Proprietary	300-400	150-200
RREL Base Catalyzed (BCD)	Bench*	Bench	Bench	>99		None	Available	250-300	20
Dechlor/KGME			Field	<99	Dewatering	Effluent, residual	Proprietary	N/A	200
Eco Logic Thermal Gas-Phase Reduction	Field*	Bench	Field	25-60	Screening	None	Proprietary	400-600	40-50
Thermal Desorption									
ReTeC Thermal Desorption	Bench		Bench	In testing	Screening	Air emissions, effluent	Proprietary	250-700	40-60
Soil Tech Anaerobic Thermal Process	Commercial		Commercial	>99	Screening	Effluent, residual	Proprietary	190-300	100-500
Thermal Desorption/UV Destruction		Field		>99		Air emissions, effluent	Proprietary	N/A	N/A

Note: N/A = Estimates are not available.
* = Bench testing results are available as part of this study.

(Continued)

Table 4 (Concluded)

Process	Previous Testing of			Efficiency %	Pretreatment Requirements	Posttreatment Requirements	Availability	Estimated Costs $/yd^3	Process Rates yd^3/day
	Sediments	Dioxins	PCBs						
Extraction Processes									
B.E.S.T. Solvent Extraction	Field*	Bench	Field	>99	Screening	Residual	Proprietary	170-190	75-100
Low Energy Solvent Extraction - ART, Inc.	Bench		Bench	<99		Effluent	Proprietary	150-200	150-200
CF Systems Propane Extraction	Field		Field	<99	Screening	Effluent residual	Proprietary	200-500	N/A
Immobilization									
Asphalt and lime addition		Bench	Field	N/A	Dewatering	Containment, monitoring	Available	20-30	N/A
International waste technologies, chemical fixation		Bench	Field	N/A		Containment, monitoring	Proprietary	130	N/A
Portland cement and pozzolan	Laboratory		Bench	N/A		Containment, monitoring	Available	50-100	N/A
Radiant Energy Treatments									
X-Ray Treatment	Conceptual	Conceptual	Bench	>99	Slurry	None	Proprietary	N/A	N/A
Thermal Treatments									
Rotary kiln incinerators	Bench*	Commercial	Commercial	>99	Drying	Air emissions	Available	350-500	100
Fluidized-bed incineration		Field	Commercial	>99	Drying	Air emissions	Proprietary	290	130
Infrared incineration		Bench	Field	>99	Drying	Air emissions	Proprietary	350-800	85-150
Supercritical water oxidation		Bench	Field	>99	Grinding, Slurry	None	Proprietary	500-850	<100
Vitrification	Bench	Bench	Field	<99	Dewatering	Air emissions, confinement monitoring	Proprietary	450-500	80-120

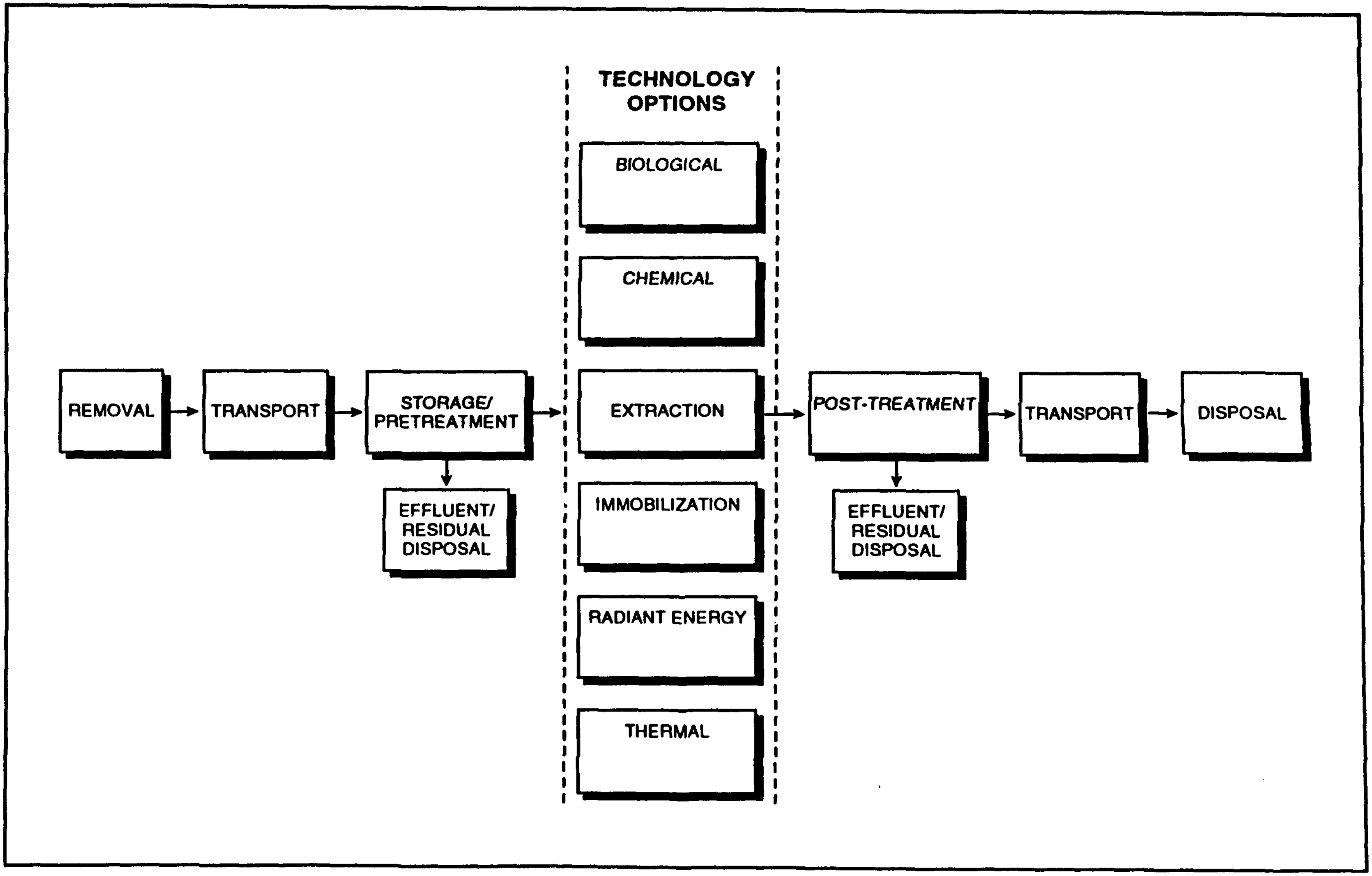


Figure 3. Remediation alternatives for contaminated sediments

For each of the treatment options, the following information is presented:

- Level of previous testing.
- Removal or destruction effectiveness of the process.
- Pretreatment requirements.
- Posttreatment requirements.
- Availability of the process.
- Estimated cost of the treatment.
- Estimated processing rate.

Most of the processes considered have been applied to sediments or to the contaminant of concern (dioxins) in a soil or sludge matrix. Those technologies that have been included and that have no track record with either contaminant or sediment treatment are in the conceptual stages of development and show promise in their applicability.

Level of Previous Testing

The level at which each technology has been successfully demonstrated is noted as either full-scale, an onsite pilot-scale (or field-scale) demonstration, bench-scale, or laboratory-scale. Full-scale operations are those carried out routinely at hazardous waste facilities or at superfund sites. The volumes of material procured are generally greater than pilot-scale demonstrations, and it is assumed that the process can be scaled up to handle large volumes of contaminated sediments and still remain commercially viable.

Pilot-scale demonstrations are intended to simulate the physical as well as chemical parameters of a full-scale process and "the pilot-units are operated in a manner as similar as possible to the operation of the full-scale system" (USEPA 1988). The contaminated material to be treated should be from (or representative of) the actual site and be identical to material expected to be treated with a full-scale system.

Bench-scale testing "usually is performed in a laboratory, in which comparatively small volumes of environmental or site-specific material are tested for the individual parameters of a treatment technology. These tests are generally used to determine if the chemistry of the process works and are usually performed in batch with treatment parameters varied one at a time" (USEPA 1988).

Laboratory-scale studies involve still smaller volumes of environmental samples, surrogate compounds, or pure substances to test the concepts of the treatment.

Effectiveness

The effectiveness of a treatment is defined as that fraction of the contaminant in the feedstock that is destroyed by the process or removed via emissions of effluents (in which the contaminant is concentrated and eventually recovered for further treatment or controlled disposal). In the case of immobilization processes, the contaminant is neither destroyed nor removed; a measure of efficiency is the percentage reduction of contaminant measurable in the leachate before and after treatment.

Because of the lack of experience in treating dioxins, and the uncertainties associated with extrapolating results from different feedstock materials and from PCB treatment results, effectiveness for the treatment options in Table 4 are indicated as either greater than 99 percent destruction or removal after treatment (>99) or less than 99 percent (<99).

Pretreatment Requirements

For all the treatment processes assessed, a land-based CDF generally is a prerequisite because sediment removal rates of commercial dredging operations are significantly greater than treatment rates. For small volumes of contaminated material, temporary storage prior to treatment could be provided by retaining the dredged material in a barge(s) instead of transferring it to a CDF. Normally, barge capacities range from 2,000 to 6,000 yd^3; and for small hot spot cleanups, one or two barges may be sufficient to hold the contaminated sediment prior to treatment.

The most common pretreatment involves altering the water content of the sediment to increase the efficiency of the treatment in question. Hydraulic dredging results in the dredged material containing about 70 percent water, whereas mechanical dredging results in material with little change in water content from that of the in situ sediment—approximately 50 percent. The majority of technologies considered can treat material with this water content with little impact on process efficiency. The pretreatment requirements listed in Table 4 are based on the assumption that the water content is similar to that of the sediments before dredging (i.e., 50 percent).

Some treatments require further reduction of water content, or drying to 10 percent water or less, for handling and maintaining process efficiencies. Other processes require the addition of water to create a pumpable slurry of feed material. Some processes require material screening or grinding of oversized material to maintain consistent particle size within the feed material. Screening of larger particles also effectively reduces the volume of material to be treated.

Posttreatment Requirements

Some treatment processes are capable of destroying dioxin and PCB contaminants by breaking them down into carbon dioxide, water, and simple nontoxic hydrocarbons. These processes do not create residues that require further treatment. Other technologies separate or extract the contaminants from the sediment, creating air streams, liquid streams, or solid residuals containing high concentrations of the contaminant or its byproducts. These streams must then undergo further treatments or be disposed of at appropriate disposal facilities.

The focus of the treatment options addressed in this study is organic contaminants, specifically dioxins. If heavy metals are also present as contaminants, they will generally remain bound to the sediment particles or, in some cases, dissolved in the liquid or gaseous streams. In these situations, further media-specific treatments may be necessary before disposal.

Availability

If the treatment has been developed as a patented process or if it is available from only one developer or vendor, it is classified as proprietary. Alternatively, if the treatment is available from a number of different suppliers or is sufficiently generic, such that it can be employed without proprietary expertise, it is then classified as available.

Costs

The purpose of this section is to compare preliminary cost estimates for removal, treatment, and disposal of contaminated dredged material. The cost analysis is based on 17 comprehensive cost categories for which general descriptions are provided below. Cost categories 1-11 will have different requirements depending upon the selected technology. These categories were chosen to provide a representative description of costs for the alternative treatment technologies addressed in this study. This will provide a common basis for cost comparisons between different technology types.

Because of characteristic differences between technologies, it is not appropriate to apply all cost categories to all technologies. In addition, because of the varying stages of development of different process technologies, cost estimates were not available for all categories listed. These cost analyses assume that all treatments are to be carried out at a land-based temporary storage facility (TSF) into which the contaminated sediments will be placed. The cost analysis for full-scale implementation was considered for three dredged sediment volumes of 10,000, 50,000, and 100,000 yd^3 and reported in 1992 dollars using present value calculations.

Present Value Calculations

Present value calculations of the costs were based upon an annual discount rate of 4.4 percent, which was the interest rate of the 3-year U.S. Treasury Constant Maturities (Federal Reserve 1993). This annual discount rate was converted to a monthly discount rate of 0.37 percent. Processing times were calculated for each technology for treatment of 10,000, 50,000 and 100,000 yd^3. The time depended on the processing rates of the technologies.

With relation to time, costs can be divided in three categories.

a. *Costs at the Beginning of the Project.* These include Dredging, Transportation to confined disposal facility (CDF), Construction of CDF, Site Preparation, Permitting and Regulatory Requirements, Pretreatment of Wastes, and Start-up and Fixed Costs. These costs are considered to occur before the processing begins and are not discounted.

b. *Costs at the End of the Project.* These include Site Demobilization and Cleanup costs. These costs are discounted for the processing period.

c. *Costs During the Project.* These include all the other costs that are not part of *a* and *b*. These costs are considered to be occurring throughout the treatment. Present value of these costs were calculated using *periodic payment formula* with uniform monthly costs.

Consideration was given to both a permanent and a temporary (in the order of a few years) treatment installation. For the purposes of these preliminary cost estimates, a scenario using a temporary installation was chosen on the basis of treating only relatively small volumes of contaminated "hot spots" instead of very large volumes of less contaminated sediment. This choice was influenced by the relatively high unit costs of all the treatment technologies. Leasing land for the TSF and treatment plant was assumed to be more realistic than purchasing land under these assumptions. Since the same assumption is made for all treatment technologies, the relative costs of the treatments will not change. However, the cost difference between treatment alternatives and disposal alternatives may change if a permanent installation requiring land purchase is chosen.

These estimates are, of necessity, approximate, and they provide a guide to the relative costs of different options. Many factors prevent precise costs from being developed for each process. Extrapolation of costs from laboratory and bench-scale studies is not precise. Costs are influenced substantially by overall volumes of material, throughput capacity of the treatment unit, contaminant concentrations, target cleanup levels, contaminant matrix interference, caloric content of the feed material, and other factors. Cost estimates are generally based on the treatment costs of PCB contamination of sediments because very little remediation of dioxins has occurred to date.

The cost categories described below, are based on the 12 line items commonly used for the economic analyses of treatment technologies performed under the SITE program.

a. *Site Preparation.* Includes site design and layout, surveys and site logistics, legal searches, access rights and roads, preparation of support facilities, decontamination facilities, utility connections, and auxiliary buildings.

b. *Permitting and Regulatory Requirements.* Includes following regulatory procedures, obtaining permits, and conducting public meetings.

c. *Capital Equipment.* Broken out by subsystems including major equipment items, process equipment, residual handling equipment, and associated equipment depreciation.

d. *Start-up and Fixed Costs.* Broken out by categories including installation, mobilization, and preoperational testing.

e. *Pretreatment of Waste.* Includes required waste pretreatment before processing. In some cases, pretreatment costs are included in the processing costs.

f. *Labor Costs.* Includes supervisory and administrative staff, professional and technical staff, operations personnel, and clerical support.

g. *Consumables, Supplies, and Utilities.* Includes utilities such as fuel and electricity, supplies, and raw materials.

h. *Effluent Treatment and Disposal.* Includes wastewater and offgas treatment and disposal.

i. *Monitoring and Analytical Costs.* Includes monitoring design, initiation, operations, and laboratory analyses.

j. *Maintenance and Repair Costs.* Includes maintenance, labor, and materials.

k. *Site Demobilization and Cleanup.* Including shutdown, site cleanup and restoration, permanent storage costs, and site security.

These planning level cost estimates were based on information supplied by the vendors and USEPA SITE Program *Application and Analysis Reports*. These estimates are standard budget estimates to commercial customers for use in planning and scoping. Only these cost categories, which are directly related to the treatment process, are considered in Table 4.

The estimates for site preparation, permitting and regulatory requirements, pretreatment of waste, effluent treatment and disposal, monitoring and analytical costs, maintenance and repair costs, site demobilization, transportation to TSF, construction of TSF, and land lease are the least accurate. Cost estimates for capital equipment, start-up and fixed costs, labor costs, consumables, supplies and utilities, dredging, transportation of residual wastes to landfill, and disposal in a rubble landfill are derived from previous work on similar projects and from extrapolation from pilot studies and can be considered the most accurate. The American Association of Cost Engineers defines three types of estimates: order-of-magnitude, budgetary, and definitive. These estimates would fall into the order-of-magnitude category. Vendor-supplied costs derived from SITE demonstrations are generally estimated with an accuracy of -30 to +50 percent, with the caveat that the actual range on the accuracy may be significantly wider. Because very little cost estimating of dioxin-contaminated material remediation has been carried out, the costs supplied in this report are only order-of-magnitude, with an estimated accuracy of one half of that estimated by the vendors, that is, from -60 to +100 percent.

Open-water and capping disposal option costs estimates are better defined and more accurate because of extensive use and the small number of components involved. The costs associated with the following steps are considered constant to all remedial technologies in treating contaminated dredged material.

a. *Dredging.* The costs associated with dredging are the best known and most easily identifiable. This is primarily due to the extensive experience developed by the Corps over the past 100 years of navigation construction and maintenance. The average costs of dredging and unrestricted aquatic disposal in the New York/New Jersey Harbor area is approximately $6 to $7.50 per cubic yard for contaminated sediments. For this study, a dredging and transportation cost of $7.50 per cubic yard is used.

b. *Transportation to TSF*. The costs of transportation of dredged sediment to a land-based TSF is about $10 to $15 per cubic yard for dredged material, assuming transport is by truck and the round trip distance is 20 miles.[1]

c. *Construction of TSF*. Dredging can be most efficiently performed at a rate that exceeds the rate that the materials can currently be treated by any of the technologies. A storage facility would need to be constructed to stockpile the dredged materials as they are dredged and before they are treated. The costs for CDFs are reasonably well known. For this study, it is assumed a simple earth and dike temporary CDF, with a simple clay or plastic liner with a small dewatering system, would be constructed as the storage component of the TSF. The estimated cost to construct a temporary storage facility was derived from the construction cost of land-based CDFs built in the Great Lakes region by USACE.[2] The average costs per cubic yard for the three sizes (10,000, 50,000, and 100,000 yd^3) were estimated using appropriate cost estimates converted to 1992 dollars. The Great Lake CDFs are simple unlined CDFs.

   The average costs to construct 10,000-, 50,000-, and 100,000-yd^3 CDFs are $15, $10, and $9 per cubic yard, respectively. These costs are at least doubled to construct lined CDFs in New York Harbor. To construct 10,000-, 50,000-, and 100,000-yd^3 capacity CDFs is estimated to cost about $30, $20, and $18 per cubic yard, respectively.

d. *Land Lease for TSF*. It is assumed that the costs for land lease are about $7,000 per acre per year.

e. *Transportation and Disposal of Concentrated Residual Materials*. The costs assume a 2,000-mile round trip to a hazardous waste landfill site, $4 per mile per 20-ton truck, and $200 per ton for disposal.

f. *Transport and Disposal of Treated Sediments*. In all examples, it is assumed that the decontaminated sediments meet all regulatory requirements for solid waste landfill disposal. The cost of transfer by truck of the treated material from the treatment facility to the landfill, for a 40-mile round trip is $4 per mile per 20-ton truck. The disposal cost at the landfill is assumed to be $58 per cubic yard.

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[1] Personal Communication, January 1993, Jan A. Miller, Environmental Engineer, USACE North Central Division.

[2] Personal Communication, February 1993, Jan A. Miller, Environmental Engineer, USACE North Central Division.

Processing Rates

As with costs, processing rates are influenced by sediment characteristics and contaminant concentrations. Because of the unavailability of cost data for treatment of dioxin-contaminated soils or sediment, the rates listed are estimates based on bench- and field-scale test results or on full-scale application on material other than sediments and on contaminants other than dioxins. Rates were frequently quoted as tons/day or pounds/hour. For consistency, these rates have been standardized to cubic yards/day by assuming 1.16 tons per cubic yard (1.4 gm per cubic centimeter) as an approximation of typical sediment density characteristics. A utilization rate of 80 percent was assumed for all treatment processes.

Public/Community/Acceptance

Public concern over the location and operation of facilities handling potentially toxic materials is an important factor in selecting a specific technology for implementation. Although not quantifiable, public concerns will create significant impacts to the siting and operation of any remediation processing facility. The major issue is the public's perception of dioxin as a high risk to human and ecological health. Significant negative reaction can be expected to be generated by plans to establish any type of dioxin-contaminated material treatment operation, regardless of the planned site location.

Another issue that will probably be raised by the public will focus on the type of proposed technology to be established at the treatment site. Some technologies are viewed less favorably by the public, irrespective of the material being processed. Processes involving hazardous conditions, such as high temperatures, high pressures, or those requiring hazardous chemicals in the process train will be less acceptable than those processes that can be carried out under normal atmospheric conditions with nonhazardous additives. Also, technologies that produce large volumes of effluents or air emissions will be viewed less favorably by the public.

Permitting and Regulatory Requirements

Special site considerations and the type of waste to be treated can strongly influence permitting and regulatory costs. The type of treatment technology can also influence permitting procedures; those processes that destroy contaminants by conversion to nontoxic substances and that do not use hazardous materials in the treatment process are more likely to obtain permits with less effort. However, because the target contaminants are dioxins, regulatory requirements and permitting may be a significant factor in terms of time and money. These costs are estimated to be $100,000 for these preliminary comparisons.

Summary of Treatment Options

The six treatment technologies, under which the processes have been categorized, are described in the following subsections.

Biological technologies

Biological treatment of organic contaminants in sediment involves the cultivation of native organisms or the addition of contaminant appropriate microorganisms and generally requires strict environmental controls, including temperature, moisture, oxygen content, acidity, and overall organic content. Bioremediation refers to the transformation of contaminants into less complex and less toxic compounds by naturally occurring microbes, by enzyme systems, or by genetically engineered microorganisms. Several naturally occurring organisms have been identified as having the ability to break down chlorinated substances, including dioxins (U.S. Congress 1991). The major difficulty encountered is establishing and maintaining the microorganism in contact with the dioxin molecules, which are tightly bound to the soil particles. This problem is exacerbated in the case of sediments where the contaminant concentrations are very low in the beginning; contact between microbes and dioxin or PCB molecules is limited; and in the case of high organic carbon content, the microbes will generally favor the more easily degradable matter over the contaminants. Processing times are much longer than other technologies; several months or years may be required to complete the biodegradation process.

Considerable bioremediation laboratory and field work has been carried out on wastewater, groundwater, and soil; but few report contaminant reductions greater than 90 percent (U.S. Congress 1991). Aerobic bioremediation has been demonstrated in bench-scale testing for PCB removal from sediments from the New Bedford Harbor Superfund site. Results suggested that greater reductions occurred in some PCB isomers than others with overall reductions being much less than 90 percent (Averett et al. 1990). Because bioremediation of dioxins in sediments have not proceeded beyond laboratory testing, no cost data are available.

Anaerobic treatments have been widely used to degrade organic contaminants in wastewater. Bench-scale testing of dredged material by anaerobic digestion indicates that only low effectiveness can be achieved (Averett et al. 1990). Recent studies in the Hudson River with PCB-contaminated sediments indicate that natural dechlorination occurs in the sediments under anaerobic conditions and that the products of this dechlorination can be degraded by aerobic processes (Jafvert and Rogers

1991). USEPA investigation of sediment from the Ashtabula and Saginaw Rivers shows that the addition of more PCBs to the sediment stimulates further dechlorination. A pilot-scale confined treatment facility with a capacity of 2,500 yd^3 was constructed in 1990 to study the effectiveness of using enhanced anaerobic biodegradation for the treatment of sediments removed from the Sheboygan River (Jafvert and Rogers 1991).

A conceptual plan, based on a sequence of anaerobic and aerobic treatments, for bioremediating sediment volumes between 10,000 and 100,000 yd^3 has been described for Sheboygan River-contaminated sediments.[1] A TSF with sufficient capacity to hold both the untreated and treated sediments is specified. The contaminated sediments are hydraulically dredged, transferred to the TSF, and amendments added to enhance the anaerobic dechlorination process. After 1 year, 10 to 25 percent of the anaerobic sediment is transferred to slurry bioreactors, appropriate amendments are introduced, and the slurry is aerated for a period of 1 to 2 months to further dechlorinate the contaminants. This portion is returned to the TSF where it is consolidated and dewatered. More anaerobic sediment is then transferred to the bioreactors for aerobic treatment, and the process is continued until all the contaminated sediment is treated. Dewatering is then completed, and the material is capped. The time required to complete the treatment is estimated to be between 2 and 3 years. The overall PCB removal rate is expected to be 50 to 60 percent.[1]

Chemical technologies

Two potential methods are identified for the chemical destruction of dioxins in sediments. Dechlorination is a process designed to chemically alter hazardous chlorinated molecules through the substitution of chlorine atoms by other atoms, generally hydrogen, resulting in a simpler, nontoxic hydrocarbon. A thermochemical reduction involves the reaction of hydrogen with chlorinated organic compounds at elevated temperatures to produce a dechlorinated compound, water, and hydrogen chloride.

Dechlorination. Three different dechlorination options show some potential for the treatment of dioxins in sediments. The major difference between them is the type of reagent used. The APEG-PLUS process (Galson Remediation Corp., East Syracuse, NY) has been developed from earlier research on polyethylene glycol (PEG) and molten sodium or potassium mixtures that break down the carbon-chlorine bonds of PCB and dioxin molecules. The proprietary reagent consists of potassium hydroxide, PEG, and dimethyl sulfoxide. Results on laboratory and bench-scale tests of contaminated soils show that this reagent is most effective on dioxins, followed by PCBs and pentachlorophenol (U.S. Congress 1991).

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[1] Memorandum, 1993, D. Hughes, Hughes Consulting Services, Syracuse, NY.

The reagent is mixed with soil or sediment to form a slurry, and the chlorinated organic contaminants are extracted from the soil particles. The slurry is heated to 150 °C, and the chlorine atoms of the dioxin molecule are replaced by PEG to form water-soluble glycol ether. This substance can be washed from the soil and easily degraded into nontoxic materials (U.S. Congress 1991). Both the reagent and wash water are recovered and can be recycled into the treatment process several times. The APEG-PLUS process has been evaluated at several contaminated sites for the remediation of dioxins and PCBs. Because the reagent mixtures are hygroscopic and exposure to water deactivates the reagent, this reagent type has not been applied to sediment treatments.

Processing rates of 200 tons/day have been achieved with PCB-contaminated soils with a destruction efficiency of greater than 99.9 percent. Cost estimates are dependent on many factors, including the nature of the soil, the treated volumes, the site characteristics, and the level of cleanup required. Estimates for one PCB/soil treatment were given as over $300 per cubic yard (U.S. Congress 1991).

KGME is another proprietary dechlorinating reagent (Canonie Environmental, Inc., Colorado) that is reported to require less reagent, to be less expensive, and to be more tolerant of the moisture content of the mixture. Successful laboratory-scale tests have been carried out on contaminated soils, and a USEPA SITE program demonstration of this process is in the planning stages (USEPA 1991a). However, this process also has not been applied to sediment cleanup.

The third process, Base-Catalyzed Decomposition (BCD) Process (USEPA RREL, Cincinnati, OH), uses lower cost basic reagents such as sodium bicarbonate and sodium hydroxide, instead of potassium hydroxide, and does not require PEG as a reagent component. This process uses reagents in concentrations as low as 1 to 5 percent of the treatment matrix, requires significantly less time for treatment, eliminates the need to recover and recycle the reagent, and achieves complete dechlorination of the contaminants (Rogers, Komel, and Sparks 1991). The process is not as sensitive to water content, although the higher the water content, the greater the amount of reagent required. This implies BCD is more applicable to sediment remediation than the earlier processes.[1] This process is one of the selected technologies presently undergoing bench-scale testing with New York Harbor sediments, and a full description and assessment of the process are presented in Chapter 3.

Because of the less expensive reagents and faster reaction time, estimated treatment costs of BCD are less than the other dechlorination processes. USEPA projections put costs at approximately $250 to $300 per cubic yard. A unit capable of processing 20 to 40 yd^3/day is presently in use, processing PCB-contaminated soils.

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[1] Personal Communication, 13 November 1992, Charles Rogers, USEPA RREL, Cincinnati, OH.

Thermal reduction. This is a thermochemical reduction process that can treat wastes containing substantial amounts of water, including sediments, landfill leachates, and tank and lagoon sludges. The principle of treatment is based on the reduction of halogenated organic compounds at temperatures of about 1,000 °C. A hydrogen atmosphere acts as the reducing agent, producing smaller lighter hydrocarbons. The reactions are enhanced by the presence of water, which combines with the lighter hydrocarbons to form carbon monoxide and hydrogen. The absence of free oxygen in the process inhibits oxidation of organic compounds and prevents the creation of dioxin and furan compounds (USEPA 1991a).

One proprietary process (ELI Eco Logic International, Inc., Rockwood, Ontario) has been successfully tested at a field demonstration on PCB-contaminated sediments (Eco Logic International, Inc. 1992a). This is another of the selected technologies undergoing evaluation, and a full description and assessment of the process is presented in Chapter 3.

Pretreatment of sediments is normally not necessary. The optimum water content is 50 to 70 percent, which is typical of dredged material. The reaction products of simpler hydrocarbons are nontoxic and do not require further treatment. Costs have been estimated by the developer to range between $400 and $600 per cubic yard for treatment of sediments containing resistant chlorinated organics (PCBs, PCDDs, and PCDFs (U.S. Congress 1991)). A 20-yd^3/day mobile unit has been operational in field demonstrations, and 80 to 100 yd^3/day would be the probable capacity of the next unit produced.[1]

Extraction technologies

Extraction involves the separation of contaminants from soils or sediment by dissolution with solvents. The solvent is later treated to remove, destroy, or collect the contaminant and is recycled into the treatment if possible. Two treatment processes have been field tested with sediments contaminated with PCBs, and a third has undergone bench testing.

The Basic Extractive Sludge Treatment (B.E.S.T.) solvent extraction process (Resources Conservation Co., Ellicott City, MD) separates contaminants into three fractions: oil, water, and solids. A combination of tertiary amines, usually triethylamine (TEA) is mixed with the contaminated material at temperatures below 20 °C. Prior to mixing, the material is screened to remove debris; sodium hydroxide is added to enhance the conservation of the TEA by raising the pH of the mixture. The organics are then separated from the solids. Inorganic contaminants remain with the solids, which may then require further treatment. The solvent mixture is heated, separating the water. The water is further processed to strip any

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[1] Personal Communication, 4 November 1992, Jim Nash, Eco Logic International, Inc., Rockwood, Ontario.

residual solvent for recycling and is then typically discharged to a wastewater treatment plant. The solvent is decanted and stripped of the condensed organic contaminants, which are now ready for final disposal or treatment (USEPA 1991a).

Bench-scale testing of the B.E.S.T. process on PCB-contaminated sediments has shown removal efficiencies as high as 99 percent (Averett et al. 1990). Cost estimates vary from $100 to $400 per cubic yard (USEPA 1993). This extraction process has been selected as one of the four treatment options for New York Harbor dioxin-contaminated sediments. Bench-scale test results are reported in Chapter 5. The process is described and evaluated more completely in Chapter 3.

Another extraction process (developed by C. F. Systems Corp., Woburn, MA) uses compressed liquified propane as a solvent to dissolve organics from contaminated soils and sludges. This process can be applied to high boiling point complex organics such as PCBs and dioxins (USEPA 1991a). The solvent is mixed with the waste stream under normal temperatures and high pressures. Typically, more than 99 percent of the organics are extracted after the third pass though. The solids and water are separated. The mixture of solvent and organics is separated by vaporizing the propane, which is then recycled. The remaining concentrated contaminants are collected for disposal or further treatment. Estimated costs for PCB cleanup of sediments are $200 to $500 per cubic yard (Averett et al. 1990; USEPA 1991a).

A USEPA SITE Program demonstration of this process was conducted on sediments from New Bedford Harbor. PCB extraction efficiencies of between 72 and 92 percent were achieved for sediments containing PCB concentrations ranging from 350 to 2,575 ppm. Residual PCB concentrations were between 40 and 200 ppm, the highest concentrations corresponding to the highest pretreatment concentrations (USEPA 1990b).

A third option is a proprietary low-energy solvent extraction process (Enviro-Sciences, Inc., and ART International, Inc., Randolph, NJ) that uses common organic solvents, including acetone and kerosene, to extract the organic contaminants from the soil matrix and then uses another solvent to strip and concentrate the contaminant. The first solvent is hydrophilic and leaches the contaminants along with the water film surrounding the solid particles. The contaminants are then concentrated in a hydrophobic solvent, which is then removed from the process. The clean solids are then separated and the hydrophilic solvent recycled. The process operates at ambient conditions. Optimum water content for treatment is reported at 30 to 50 percent (USEPA 1991a).

The process has completed bench-scale testing on PCB-contaminated sediments. A 200-yd^3/day mobile unit is now in operation. Processing costs for PCB reduction are estimated at $150 to $200 per cubic yard, which includes incineration costs of the concentrated contaminants.[1]

Immobilization technologies

Contaminant immobilization techniques focus on limiting the solubility or the mobility of contaminants present in the medium. In situ (i.e., subaqueous) immobilization is technically infeasible at present and undesirable in the case where the purpose of dredging is to maintain navigable depths in channels and harbors.

Immobilization treatment options with possible application to dioxin-contaminated sediments fall into three categories; the most common is stabilization/solidification. Vitrification is another approach that is discussed under thermal technologies because the process involves pyrolysis. The third option is confined disposal, either land-based or aquatic, in which the material is permanently contained within a site. Water is necessary to improve the efficiency of mixing reagent with the sediments or soils. However, excessive water content, such as associated with hydraulically dredged slurries, must be reduced for effective treatment. Posttreatment considerations common to all processes are the continued containment of the stabilized or solidified mass and the monitoring requirements associated with such a confinement facility.

Stabilization/solidification processes involve the physical bonding of contaminants to the medium through the addition of various cementing agents. In laboratory testing of dioxin-contaminated soils, mixing emulsified asphalt and lime with the soil appeared more effective than other additives. Dioxins were not detected in analyses of the leachate. Estimated treatment costs ranged from $20 to $30 per cubic yard (U.S. Congress 1991). Portland cement is a common bonding agent in these processes, as is pozzolan (fly ash, kiln dust, etc.); the combination of the two improve the strength, handling characteristics, and chemical resistivity of the concrete-like product. Laboratory testing of sediments indicates that leachabilities of PCBs were reduced by one to two orders of magnitude. Estimated costs based on testing of a contaminated sediment matrix were $55 per cubic yard (Averett et al. 1990).

The addition of proprietary compounds (organophilic clays, soluble silicates, emulsifiers, and sorbents) to the cement mixture has the potential to improve performance by promoting chemical bonding between the contaminant and the cement matrix. International Waste Technologies (Wichita, KS) has bench tested an in situ chemical mixing and fixation process on PCB-contaminated soils. The process covalently bonds contaminants to

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[1] Personal Communication, 12 November 1992, Barry Rugg, ART International, Inc.

inorganic polymers, creating a cement-like monolith. The addition of the mixture increases the density of the soil by 20 percent and the volume by a little less than 10 percent. European bench-scale testing of soils and sludges demonstrated that maximum dioxin concentrations detectable in the leachate after treatment were 10 pptr (U.S. Congress 1991). Compressive strength increased and porosity and permeability decreased over time, indicating the potential for long-term durability. Based on bench-scale testing, costs are estimated to be $130 per cubic yard. Processing rates were not given.

Radiant energy technologies

These are processes that incorporate photodegradation as the means of breaking down organic contaminants. Of the various conceptual methods presently available, no process has successfully demonstrated promising application to contaminated sediment beyond the laboratory testing stage. One potential process, still in the conceptual stage is an X-ray treatment (Pulse Sciences, Inc., Agoura Hills, CA) designed to treat organically contaminated soils and liquids.

A prototype unit consists of a 1.2 million electron-volt linear induction accelerator and X-ray converter that delivers a 55-nsec pulse of radiation to the waste material. Small-scale samples will be treated with dose levels of 100 kilorads to 5 megarads, depending upon composition. The higher the water content, the smaller the dose required. It is expected that complete mineralization of contaminants can be achieved without volatilization or the creation of toxic residuals.[1]

This technology has been accepted into the USEPA's SITE Emerging Technologies Program, and laboratory-scale tests of PCB-spiked materials are in progress. According to the technology developer, the process will be enhanced with a typical sediment water content of 50 percent, eliminating the need for dewatering before treatment. Destruction efficiencies are projected to be greater than 99 percent, with no by-products being formed.[1] Accurate estimates of treatment costs have yet to be developed.

Thermal treatment technologies

Thermal technologies are those that use heat as the primary treatment agent. Incineration processes, pyrolysis (in conjunction with vitrification), and supercritical oxidation are treatment types considered promising for the treatment of dioxin-contaminated sediments. A number of alternative incineration processes are considered. As part of this study, bench-scale test incineration of New York Harbor sediments has recently been

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[1] Personal Communication, 12 November 1992, R. Curry, Pulse Sciences, Inc., Agoura Hills, CA.

performed at the USEPA Incineration Research Facility (IRF) by Acurex Environmental Corporation. A bench-scale thermal treatment test unit (TTU) was used to characterize the results of incineration of the test sediments. These results will be used to determine if incineration is effective in destroying dioxins in New York Harbor sediments and to determine the impact of different operating conditions on the destruction effectiveness.

Several thermal treatment processes have been tested recently for the destruction of dioxin and PCB-contaminated materials. However, because of the high water content inherent in untreated sediments, few tests have used sediments as feedstock.

Rotary kiln incineration. Of the five thermal treatment processes that show potential for future application, only rotary kiln incineration has been fully demonstrated and used commercially for cleaning up dioxin in soil (U.S. Congress 1991). At the present time, several installations have Toxic Substances Control Act (TSCA) permits to burn PCB wastes; but none have incinerated dioxin-containing wastes because of the lack of operating permits, which for this contaminant are issued under the authority of the Resource Conservation and Recovery Act (RCRA).

Several mobile rotary kiln incinerators have been built based on USEPA research incinerator design and experience. Mobile kilns can be installed quickly at a contaminated site, avoiding the necessity of long distance transport of contaminated materials to a permanent incineration facility.

The kiln is comprised of a rotating refractory-lined cylinder. A shredder, afterburners or secondary combustion chambers, and air pollution control systems are integral parts of nearly all designs. The material waste (or dewatered sediment) is first burned to inorganic ash and gases at temperatures in excess of 1,200 °C in the rotary kiln. The ash is collected, and the gases undergo further combustion in the secondary treatment chamber for complete destruction of organic contaminants. The USEPA research incinerator has achieved destruction and removal efficiencies (DREs) of 99.9999 percent in the treatment of 2,3,7,8-TCDD, and has successfully incinerated more than 12 million lb of dioxin-contaminated soils and 230,000 lb of dioxin-contaminated liquid waste from eight sites between 1985 and 1989 (Gupta 1990).

Costs and the speed of treating sediments are directly related to the size of the incineration unit, the water content, and the caloric content of the material. Current stationary units are capable of processing between 80 and 150 yd^3/day; mobile units have reported capacities of between 40 and 130 yd^3/day, within a cost range of $450 to $700 per cubic yard (U.S. Congress 1991).

Fluidized bed incineration. This is a nonproprietary process in which heated air is forced into the bottom of a vertical refractory-lined vessel and through a perforated plate that holds a catalyst and a limestone/sand mixture. The passage of the heated air fluidizes the bed material and entrains the solid waste and limestone mixture introduced from near the top of the chamber, creating a highly turbulent combustion zone (1,400 to 1,600 °C) that ensures mixing of the waste and bed particles and complete combustion in seconds for gases and in minutes for liquids (Averett et al. 1990). Solid noncombustible materials are drawn into a cyclone separator. Exhaust gases must pass through pollution control systems (secondary combustion chambers, scrubbers, or precipitators) before release to the atmosphere. Some effluent will be created during this process. Waste-Tech Services, Inc., Golden, CO., has completed trial burns as part of a RCRA permitting process and is pursuing a TSCA permit for the treatment of PCBs. Dioxin and furan-contaminated samples were tested, and no 2,3,7,8-TCDD was detected in the treated samples (U.S. Congress 1991).

The circulating bed combustor design (Ogden Environmental Services, Inc., San Diego, CA) is a modification of a fluidized-bed incinerator system in which high-velocity air is used to circulate the solids and a bed material composed of a mixture of contaminated soils and limestone. The rapid movement of the air promotes efficient mixing of the waste at lower temperatures (800 to 1,100 °C) and without requiring a secondary combustion chamber for exhaust gases. These gases are passed through a cooler and a baghouse before release to the atmosphere. Noncombusted solids are removed via a cyclone separator after a treatment time of approximately 30 min (USEPA 1991a). DREs of 99.9999 percent have been reported for soils highly contaminated with PCBs. No dioxin tests have been reported. Daily processing rates of 130 yd^3 are estimated for the largest available unit, depending upon the heating value of the feed material (USEPA 1991a). According to Ogden officials, the cost of processing more than 20,000 tons of soil in 1991 was approximately $290 per cubic yard (U.S. Congress 1991). These processing rates are expected to be lower (and costs higher) for sediments with low organic content and will be most dependent upon water content. Dewatering appears to be a beneficial pretreatment.

Infrared incineration. A mobile infrared treatment unit developed by Shirco (ECOVA, Richland, WA) has successfully bench-tested dioxin-contaminated soils. The primary combustion chamber consists of electrically heated silicon carbide elements capable of achieving infrared irradiation at temperatures up to 1,850 °C. The partially combusted solids and gases are then treated in a gas-fired secondary chamber to complete combustion and are passed through a wet scrubber to remove particulates. The treated solids are separated from the primary combustion chamber and cooled with scrubber effluent water before being discharged. As in all incineration treatments, costs and efficiency of treatment will improve with decreases in water content of the feed material.

Bench-scale testing for dioxin treatment have achieved DREs in excess of 99.9999 percent. However, full-scale demonstrations have not been as successful because of scrubber inefficiencies. Full-scale 85- to 150-yd^3/day units are commercially available. Economic analyses of costs vary greatly, from $350 to $800 per cubic yard (U.S. Congress 1991; USEPA 1991b).

Thermal desorption. This is a method of directly or indirectly heating soil, sediment, or sludge to a temperature sufficient to volatilize organic contaminants into a purge gas, where they can be separated for treatment. The temperatures required for volatilization are lower than for incineration, and energy requirements are also less. Air is not required for combustion; in fact, most processes are carried out in inert atmospheres.

Another proprietary process (Remediation Technologies, Inc., Concord, MA) is based on the use of a hollow double-screw mechanism to move the material through the processor and to heat it by circulating a hot fluid through the augers. The fluid is a molten salt eutectic heated in an electric or oil/gas-fired heater and pumped through the augers and a jacket around the processor in a closed system.

Particles larger than 1 in. may need to be screened prior to treatment, and excess liquids may need to be removed before processing to reduce the material volume and heating requirements. Material to be treated may contain up to 60 percent water, making the process suitable for soils, sludges, and sediments. The material undergoing treatment can reach temperatures of 850 °F. An inert gas, such as nitrogen, is used to enhance the removal of volatilized organics from the thermal processor and to inhibit oxidation. The gas and the vaporized water and organic compounds are treated in a secondary offgas processor. The purge gas can be recycled, and the liquid is separated into water and oil components for further processing or disposal.

This process has extensive commercial experience with sludge treatment. A mobile demonstration unit has recently completed tests on PCB and other chlorinated hydrocarbon-contaminated sediments from the Ashtabula River in Ohio under the Assessment and Remediation of Contaminated Sediments (ARCS) program. Results from this testing are expected by mid-1993. Commercial thermal desorption units with capacities of 40 to 60 yd^3/day are in operation at various refineries throughout the country. Treatment costs range from $250 to $750 per cubic yard for refinery sludges. Organic compounds, including halogenated and nitrogenated compounds and oily sludges are treatable. Removal efficiencies for soil from a manufactured gas plant are generally greater than 99 percent for semivolatile organic contaminants (Remediation Technologies 1992).

An Anaerobic Thermal Processor (ATP) developed by Soil Tech ATP Systems, Inc. (Englewood, CO), has recently completed treatment of PCB-contaminated soils at the Wide Beach Superfund site in New York and PCB-contaminated sediments from the Waukegan Harbor, Illinois Superfund site. This is a proprietary thermal desorption process incorporating four process units or zones: preheat, pyrolysis, combustion, and cooling. Low temperature volatiles are extracted at temperatures up to 500 °F in the preheat zone, which is indirectly heated from the hot treated material exiting the processor. The material is then passed into the pyrolysis zone and is heated to 950 to 1,150 °F by treated material recycling from the combustion zone. Solid material is then conveyed to the combustion zone where the less volatile hydrocarbons are vaporized, producing hot, decontaminated inert solids. Temperatures in this zone reach 1,400 °F. The energy is supplied by natural gas burners. The vapors from both the combustion zone and the preheat zone are scrubbed and separated; light hydrocarbons can be returned to supplement the combustion fuel. The condensed liquids containing the heavier hydrocarbons are separated into water and oil components. The condensed oil containing the heavier contaminants are pumped to storage for subsequent disposal offsite (Montgomery, Rogers, and Kornel 1992; USEPA 1992a).

The average PCB removal efficiency from treated soils and sediments at the Waukegan Harbor site was 99.96 percent with residual concentrations in the treated material of 2 ppm. Furan concentrations of 102 ppb were reduced to 5 ppb. Stack emissions attained efficiencies of greater than 99.9999 percent, and no dioxins or furans were detected in the stack gases. Processing rates during the testing and treatment of 11,000 yd^3 of soils and sediments averaged over 8 tons/hour (165 yd^3/day). Processing costs were approximately $240 per cubic yard, including fixed costs (Hutton and Shanks 1992).

The sediments required dewatering before treatment; the addition of oil to the feed material may be necessary to act as the PCB solvent and condensate. Screening and crushing may be necessary if waste material particle size exceeds 2 in. The process wastewater or aqueous condensate requires further treatment once it is separated from the oil condensate. This includes filtration, oxidation, and adsorption treatments. The oil condensate, which contains the highly concentrated contaminants, up to 32 percent PCBs at the Waukegan Harbor treatment, then requires further treatment or appropriate disposal (Hutton and Shanks 1992).

This ATP unit was also used to treat 40,000 tons of PCB-contaminated soils at the Wide Beach Superfund site. This process involved a combination of thermal desorption and alkali polyethylene glycol (APEG) dechlorination. Oil was added to the vapor condensation phase and APEG added to the oil/water separation phase to dechlorinate the PCB fraction that had desorbed from the waste feed. PCB levels of 4 to 40 ppm were reduced to nondetect levels of 0.5 ppm (Soil Tech, Inc. 1992).

Another thermal desorption treatment, combined with ultraviolet (UV) radiation treatment is under development by IT Corporation. This process consists of heating the soil matrix to 560 °C to volatilize the dioxin, collecting the vapors in a solvent, and subjecting the solvent to UV radiation to decompose the dioxin molecules by photochemical reaction (U.S. Congress 1991).

Efficiencies of greater than 99.9 percent have been obtained in bench-scale testing of soils, and field testing of PCB-contaminated soils is planned. No cost estimates or processing rates are available.

Supercritical water oxidation. This is a relatively new process based on the oxidation ability of water under supercritical conditions (340 to 450 °C and greater than 218 atmospheres of pressure). Organic compounds become very soluble, and normally soluble inorganic salts tend to precipitate under these conditions. Contaminated material, liquid oxygen, and air are pumped into the heated, pressurized reactor vessel. Organic contaminants are oxidized to simpler hydrocarbons, carbon dioxide, nitrogen, and water. A solid residue of metal salts and a clean effluent of purified water are created (U.S. Congress 1991).

Although this is a batch processing treatment with potentially high processing rates, no testing beyond bench-scale for dioxins has been performed. Tests for PCB treatment have resulted in DREs greater than 99.9999 percent, and the formation of dioxins was not detected in the treated liquid. Application of this process to sediment may not be straightforward, however. Increasing the capacity of this closed system to handle the required volumes of sediments may prove difficult; the low caloric content of sediments may require augmentation by the addition of fuels such as organic matter, natural gas, or fuel oil and considerable pretreatment, including grinding and pulverizing and slurry injection into the reactor.

Costs and processing rates are estimates derived from the developers' data because no full-scale operations have been carried out. Costs are reported to be less than rotary kiln incineration at $500 to $850 per cubic yard (Averett et al. 1990) for a medium-sized unit capable of handling 20,000 gal/day. This is equivalent to less than 20 yd^3/day if the material is introduced into the system in a 20-percent solids slurry (U.S. Congress 1991).

Vitrification. This is a pyrolysis-based process in which high temperatures are created by electric currents flowing through electrodes immersed in the waste material. Organic contaminants are combusted or volatilized, and the remaining material is solidified. This treatment has been developed as a batch process using a reaction vessel and as an in situ process where electrodes are introduced into the soil. As electricity is applied, the material between the electrodes is heated to 1,000 to 2,000 °C; melting occurs and the material then becomes more conductive, facilitating the melting of surrounding material. During this process, organic contaminants are decomposed into simpler compounds of carbon, hydrogen, and chlorine; nitrates break down into nitrogen and oxygen, and inorganic soil compounds are converted to silica and alumina oxides. Volatile compounds are given off and collected and treated in pollution control systems such as scrubbers and precipitators.

Various reactor vessel-based systems are undergoing testing within USEPA's SITE program (e.g., Vortec Corp., Collegeville, PA; Retech, Inc., Ukiah, CA). These processes utilize differing methods, such as plasma arc heating and combining oxidation of the offgases. The final product is some type of glass pelletized material with low leachability and toxicity.

An in situ method with potential for sediment treatment and presently undergoing bench-scale testing within the SITE program is that developed by Geosafe Corporation, Kirkland, WA. Four electrodes are inserted into the soil or sediment to be treated, and a hood is placed over the treatment area to collect gases and water vapor resulting from the melting. These gases are treated by quenching, scrubbing, and filtering. Upon cooling, a vitrified monolith is created with a silicate glass and microcrystalline structure. (A volume reduction of greater than 25 percent also occurs.) The electrodes are then repositioned to incorporate this mass into the next melt (Hansen and Fitzpatrick 1991).

Bench-scale testing of dioxin-contaminated soils and of PCB-contaminated sediments has exceeded USEPA's efficiency requirements of 99.9999 percent DRE (U.S. Congress 1991; Reimus 1988). Water content reduction is a recommended pretreatment because of the extra energy and costs involved in volatilizing the excess water. However, some moisture enhances the conductivity of the material. Posttreatment concerns include the control of offgases from the process, the removal of the vitrified residue for reactor vessel processes, and the continued confinement and environmental monitoring of the in-ground vitrified monolith (Hansen and Fitzpatrick 1991).

No cost data for large-scale applications are available. Developer's estimates for sediment treatment, based on bench-scale testing, range from $275 to $650 per cubic yard. Costs are heavily dependent upon site-specific factors, including moisture content and electricity costs. Full-scale production for one unit is estimated at 80 to 120 yd^3/day.[1]

Treatment Technologies Selected for Further Evaluation

Thus far this chapter has reviewed a number of treatment technologies that are potentially applicable to NY/NJ sediments. The list presented in Table 4 represents results of a screening of technologies from the literature based in large part on a previous comprehensive review of existing treatment technologies for contaminated sediment by Averett et al. (1990). The next chapter of the report will review six treatment alternatives in detail, including an evaluation of treatment effectiveness, implementability,

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[1] Personal Communication, 11 November 1992, J. Hansen, Geosafe Corporation, Kirkland, WA.

and cost. Each alternative presented considers all of the steps necessary to implement the alternative, including dredging, transport, pretreatment, treatment, posttreatment, and disposal. A comparison of the treatment alternatives, as well as three disposal alternatives, will be presented in Chapter 5.

The purpose of this section is to narrow the treatment technologies presented in Table 4 to six representative technologies, which are reviewed in more detail in Chapter 3. The primary objective of this is to select process options for several different technology types that have high potential for full-scale remediation of dioxin-contaminated marine sediments. Three factors are considered for a subjective rating of the treatment technologies presented in Table 4: potential effectiveness for dioxin treatment, ease of implementation for contaminated sediment, and potential for full-scale implementation. Each technology is given a rating of high (3 points), medium (2 points), or low (1 point) for each of the three factors. Definitions for each factor and rating are provided in Table 5.

Table 5
Rating Factors for Selection of Existing Treatment Technologies for Further Evaluation

Factor	High (3 points)	Medium (2 points)	Low (1 point)
Potential effectiveness for dioxin treatment.	Destructive or removal process with >99 percent effectiveness in destroying dioxin.	Highly effective removal process requiring posttreatment of residues or destructive process with <99 percent effectiveness in destroying dioxins.	Process improving handling and disposal of sediments, but with no destruction or removal of dioxins from sediment.
Ease of implementation for sediment.	Limited pretreatment of sediments required. Applicable to wide range of water contents and sediment physical characteristics.	Sediment dewatering required for economical and/or effective processing. Fine-grain material adversely affects process.	Sediment drying required.
Level of current implementation.	Full-scale unit has been demonstrated for soils or sediments.	Pilot-scale unit is available and has been demonstrated.	Process is emerging and has only been evaluated in the laboratory.

The results of the treatment technology rating are presented in Table 6. Four of the nineteen technologies evaluated were given scores 8 out of 9 possible points, ten had totals of 7, and the remaining five were rated as 6 or lower. All of the ratings are close because a long list of technologies has already been screened once to select the more promising existing technologies. Any of the top 14 technologies have high potential for treatment of NY/NJ sediments. However, only six representative technologies will be carried forward for detailed evaluation.

Table 6
Selection of Treatment Technologies for Further Evaluation

Process	Potential Effectiveness for Dioxin Treatment	Ease of Implementation for Sediment	Level of Current Implementation	Total Score
Bioremediation	2	2	3	7
APEG-Plus	3	1	3	7
Base-Catalyzed Dechlorination	3	3	2	8
Dechlor/KGME	3	3	2	8
Thermal Gas-Phase Reduction (Eco Logic)	3	2	2	7
Triethylamine Extraction (B.E.S.T.)	2	3	3	8
Low Energy Extraction Process (ART)	2	3	2	7
Liquid Propane Extraction (CF Systems)	2	2	3	7
Asphalt/Lime Solidification	1	2	3	6
Proprietary Solidification	1	3	3	7
Portland Cement/Pozzolan Solidification	1	3	3	7
X-Ray Treatment	2	1	1	4
Rotary Kiln Incineration	3	2	3	8
Fluidized Bed Incineration	3	2	2	7
Infrared Incineration	3	2	2	7
Thermal Desorption (ReTeC)	2	2	2	6
Anaerobic Thermal Processor (Soil Tech)	2	2	3	7
Thermal Desorption/UV Destruction (IT)	3	2	2	7
Supercritical Water Oxidation	3	1	1	5
Vitrification	2	1	2	5

The six selected technologies are bioremediation, base-catalyzed dechlorination, thermal gas-phase reduction, triethylamine extraction, Portland cement solidification, and rotary kiln incineration. Although bioremediation has not been demonstrated to be highly effective for dioxins, much research is being devoted to biotreatment technology; and biotreatment offers a chance for less expensive treatment compared with chemical or thermal methods. Therefore, bioremediation was retained for detailed evaluation as an option for the future.

The three dechlorination processes all have high potential. Base-catalyzed dechlorination was selected for further analysis because it represents the most recent innovations in this type of technology and may offer cost savings by using less expensive reagents compared with the organic compounds required by Dechlor/KGME or APEG-Plus.

The thermal gas-phase reduction process actually represents a thermal desorption technology and a destructive operation for the desorbed dioxins and will be retained for further analysis. The thermal desorption/UV destruction process offers the advantage and was equally rated. Of the extraction processes, the triethylamine process was selected because it has been more extensively tested and has been executed for at least one full-scale cleanup.

Solidification processes do not remove or destroy dioxins or other contaminants, but they may reduce the leaching and bioavailability of the material, and affect both organic and inorganic contaminants. The more readily available Portland cement process was selected for detailed evaluation.

Rotary kiln incineration was selected as the thermal process for detailed review because it has been demonstrated on a full scale for dioxin-contaminated soils, and because it is readily available from several sources. The thermal desorption processes that produce side streams for posttreatment were not selected for further review because of the posttreatment requirement and their increased cost for drying wet sediment. Supercritical water oxidation, vitrification, and X-ray treatment were not rated favorably in this screening process because of their current state of development. This screening was performed to narrow the list of technologies for a more detailed review as part of this study. Future studies may well include these emerging technologies for contaminated sediment remediation.

3 Detailed Evaluation of Selected Treatment Technologies

Thermal Gas-Phase Reduction Process

This is a patented thermo-chemical reduction process (ELI Eco Logic International, Inc., Rockwood, Ontario, Canada) that can treat wastes with substantial water content, such as marine sediments, landfill leachates, and tank and lagoon sludges. The principle of the treatment is based on the gas-phase reduction of organic and halogenated organic compounds at temperatures in the range of 850 to 1,000 °C. A hydrogen atmosphere acts as the reducing agent, producing small, lighter hydrocarbons, primarily methane and ethylene. The reactions (Figure 4) are enhanced by the presence of water, which combines with methane in a reducing atmosphere to form carbon monoxide and hydrogen. The absence of free oxygen during the process inhibits oxidation of organics and prevents the by-production of dioxin/furan compounds.

Description of treatment process

As with nearly all sediment treatment technologies, dredging and transport to a treatment site is necessary. After treatment, the clean material requires final disposal, as does the effluent and residuals produced as a result of the process. Figure 5 is a flowchart of the required remediation components describing the overall treatment procedure from removal to disposal.

A detailed process schematic of the treatment is presented in Figure 6. This process has recently been expanded to a two-step treatment. The first step is a desorption process that separates organic contaminants from the sediment solids. The sediment is pretreated by screening debris before adding it to the feed hopper of the thermal desorption unit (TDU). The TDU is a bath of molten tin in a hydrogen atmosphere. Contaminated sediment is fed from the storage hopper by an auger mechanism onto the

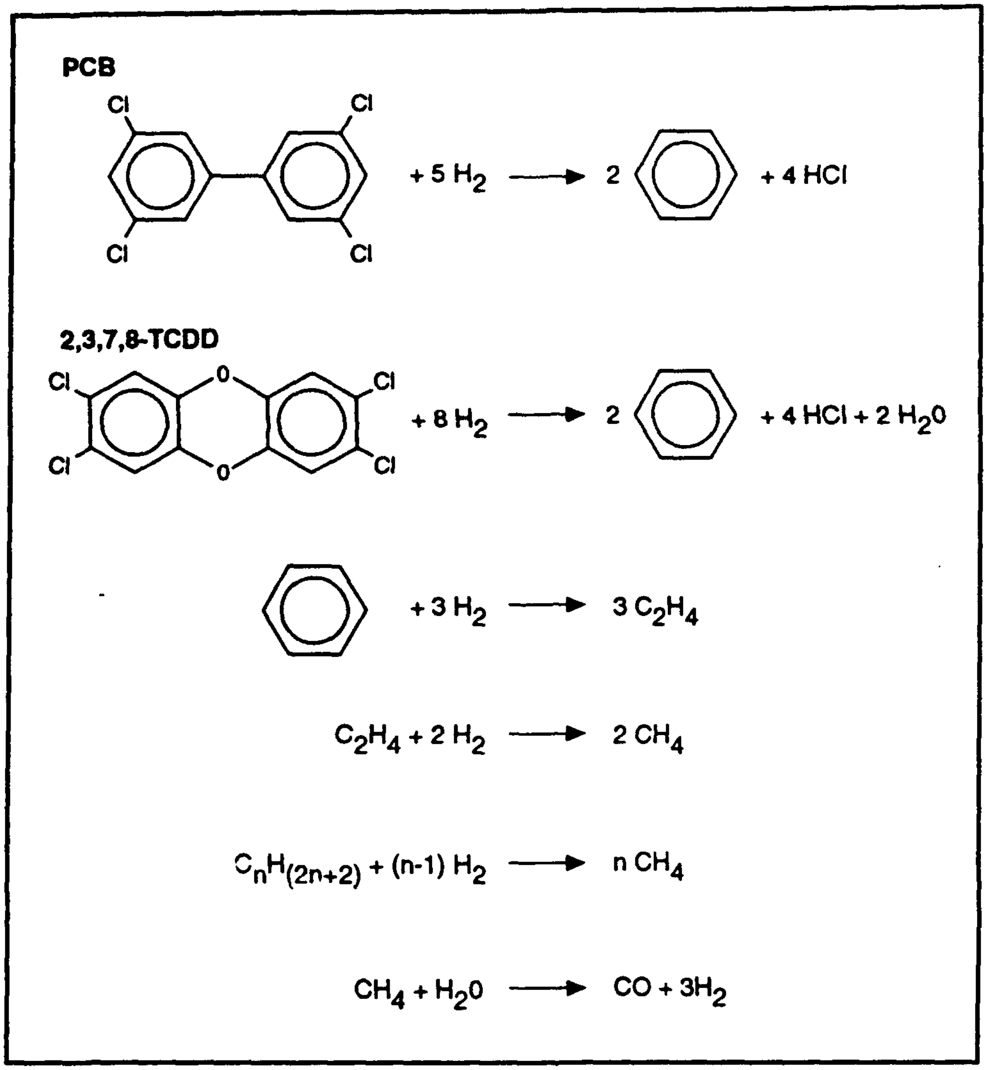

Figure 4. Thermal reduction reactions

surface of the molten tin. The waste floats on the surface because of the higher density of the tin. Slowly moving blades skim the surface of the tin, moving the waste through the TDU. Sufficient heat is absorbed by the sediment particles to vaporize (desorb) all bound organics. The vapor stream then enters the thermo-chemical reduction system for further processing. The processed sediment is transferred to a water-filled quench tank for cooling and subsequent disposal (Eco Logic 1992b). The vapor stream is injected into the reactor vessel through atomizing nozzles. A mixture of hydrogen gas and recirculation gas are heated to 500 °C in a separate gas-fired heat exchanger and passed into the reactor vessel.

This mixture circulates around a central ceramic tube and past silicon carbide heating elements, reaching temperatures greater than 850 °C by the time it passes through ports at the bottom of the ceramic tube. The gas stream flows up the center of the ceramic tube where the reduction process occurs. The reaction takes less than 1 sec to reach equilibrium.

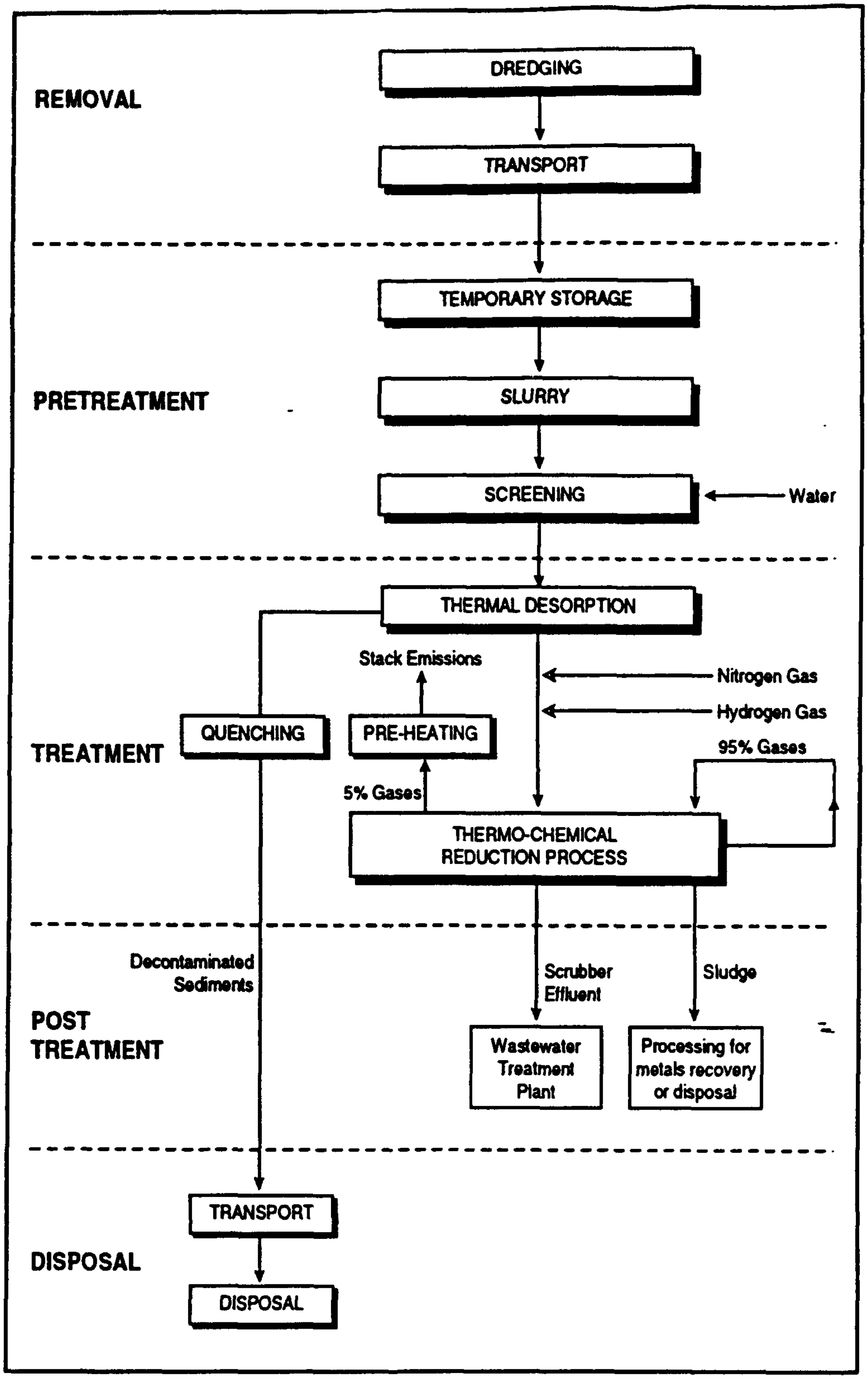

Figure 5. Eco Logic gas-phase reduction process flowchart

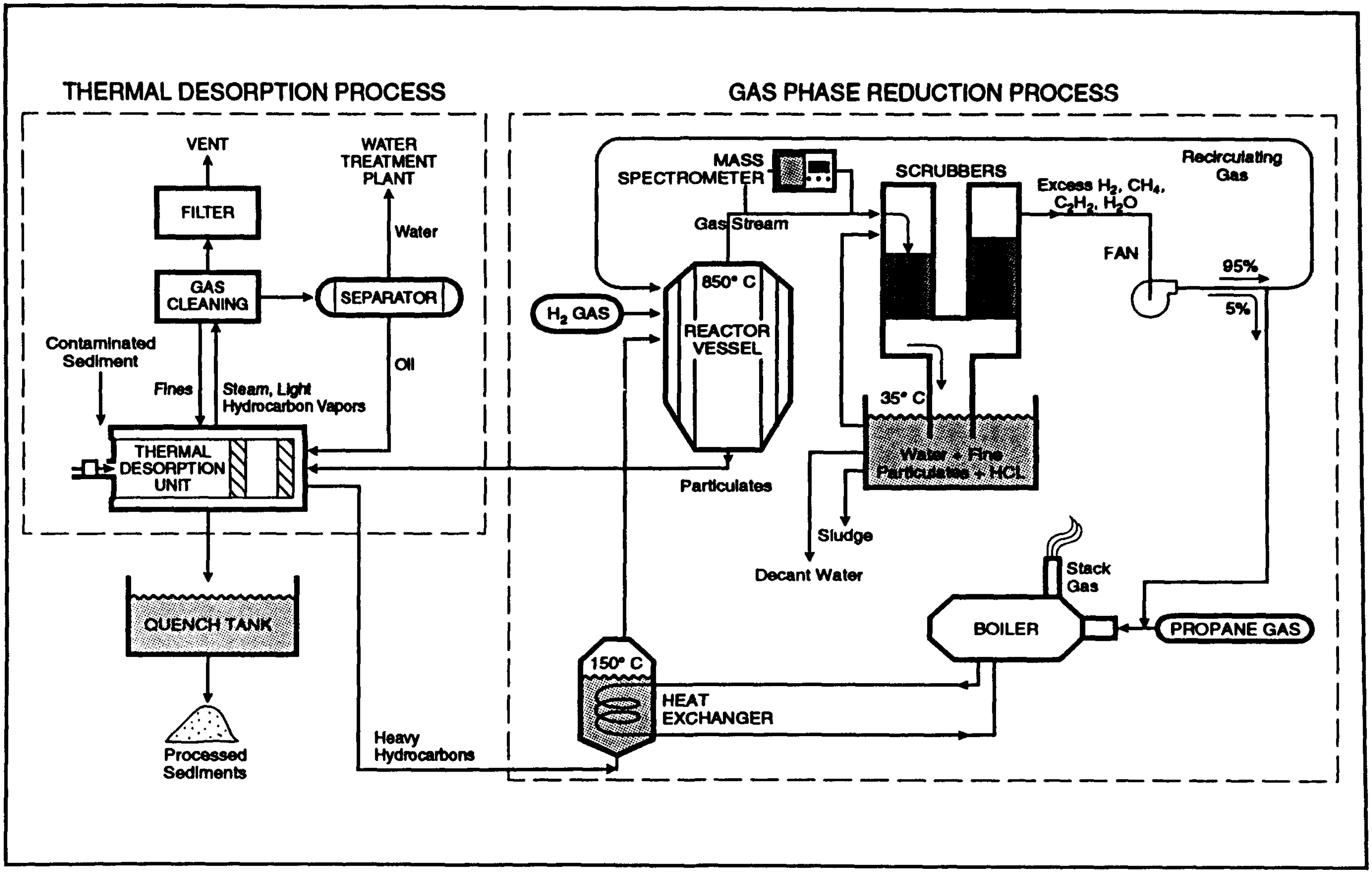

Figure 6. Eco Logic gas-phase reduction process schematic

The gases then exit through the top of the reactor and enter the scrubber unit where they are quenched by a water spray. Hydrogen chloride and fine particulates are removed by contact with the scrubber water as the gases pass through the carbon steel and polypropylene filters. Scrubber effluent is vented at the bottom of a large tank. This tank also acts as an emergency hydraulic pressure relief duct. The scrubber water is cooled to 35 °C using a heat exchanger supplied by cooling water from an evaporative cooler (Eco Logic 1992a).

Sludge and decant water from the quench tank are the only two effluent streams from the scrubber operation. Both effluents are held in tanks for analysis and storage prior to disposal. In the Hamilton Harbor, Ontario, demonstration operations, the liquid effluent, representing the largest volume of effluent from the process (equivalent to the amount of water processed with the sediment), was acceptable for discharge to conventional wastewater treatment plants. The scrubber sludge is a minor by-product, approximately 1 percent of the decant water volume. It consists primarily of lime, carbon, fine particulates, and water. From the demonstration test, it was shown that this sludge is suitable for landfill disposal; however, the associated analytical costs make it more economical to recycle this small volume back into the waste input stream (Hallett and Campbell 1991). Disposal requirements of sludge and decant water may vary and depend on characteristics of feed sediment and regulatory requirements.

The gases that exit the scrubber consist of excess hydrogen, light hydrocarbon reduction products, such as methane and ethylene, and water vapor. Approximately 95 percent of this mixture is recirculated through the heat exchanger and into the reactor vessel, and the remainder is used as supplementary fuel in the boiler. The propane-fueled boiler is used for the preliminary heating of the waste stream. The only air emissions are from the boiler in the form of stack gas. Because the fuels used in the boiler are very clean and contain no chlorine, emissions from the boiler have been shown to be insignificant (Eco Logic 1992a).

An in-line mass spectrometer continuously analyzes the treated gases. In the case of a process malfunction, or under conditions of suboptimal performance efficiency, the side stream gas to the boiler is stopped. All gas is then recirculated into the reactor vessel until the continuous analysis indicates that destruction efficiencies have again reached acceptable levels. During this time, the scrubber water may become contaminated and require treatment, but no escape or incineration of chlorinated organic compounds occurs (EcoLogic 1992a).

State of Development

To date, development of this process has included bench-scale testing of surrogate compounds, refinement of a larger laboratory-scale unit for testing actual waste samples, and construction of a mobile field unit (2.5

to 6.0 yd^3/day) for materials and component testing (Table 7) (Hallett and Campbell 1991).

Table 7
State of Development: Gas-Phase Thermal Desorption Process

Developer/ Sponsors	Scale of Tests	Material Tested	Volumes or Processing Rates	Target Contaminants	Comments	References
Eco Logic/ USACE-New York District	Bench	Sediments	<10 gal	Dioxins/Furans (PCDD/PCDF)	Sediment samples from NY Harbor	
Eco Logic/ USEPA-SITE, Environment Canada	Field	Oil/Water, Soils	2.5 to 13 cu yd/day	PCBs, TCE	Middleground Landfill Superfund Site Bay City, MI	1
Eco Logic/ USEPA-SITE, Environment Canada Environment Ontario	Field	Sediments	2.5 to 6.0 cu yd/day	Coal tar, PAH, PCBs	Hamilton Harbor	2
Eco Logic	Bench	Sediments Environmental wastes	2 lb/hr	Coal tar, PAH, PCBs, PCDD/ PCDF, Fe, ZN Wood treatments Chlorophenols	Hamilton Harbor, Thunder Bay Harbor, Sheboygan Harbor	2
Eco Logic	Bench	Pure	<1 lb	PCB, Hexachlorobenzene Trichlorobenzene 2,4-D Methoxychlor		2

Note: 1 = Hallet and Campbell (1991); 2 = Campbell and Hallett (1992).

Bench-scale testing of pure compounds, including the PCB Aroclor 1254, hexachlorobenzene, trichlorobenzene (TCB), a herbicide 2,4-D, and methoxyclor resulted in DREs of greater than 99.9999 percent. Laboratory tests with a processing rate of 2 lb/hr have been performed on sediment samples taken from the Great Lakes (Hamilton Harbor, Thunder Bay Harbor, and Sheboygan Harbor). These sediments were found to contain a range of contaminants, including PCBs at 300 ppm, dioxins and furans, PAH, TCB, coal tar, wood-treatment products, chlorophenols, and iron and zinc.

A field-scale test unit was mobilized at Hamilton Harbor, Ontario, between April and August 1991 to process harbor sediment contaminated predominantly with coal tar (PAHs). A series of characterization and performance tests of 2 to 4 hr duration and with feed rates of 1 to 5 lb/min of slurried sediments (5 to 10 percent solids) were conducted with a mobile processing unit installed onsite. This unit has the capacity of between 2 and 7 tons of solids per day. Neither PCBs, nor dioxins were a focus of the tests; however, a sample was spiked with PCB-contaminated oil. The process reduced concentrations to below detection limits, and DREs were reported as better than 99.999 percent (Eco Logic 1992a).

Bench-scale testing of dioxin contaminated sediments from New York Harbor was conducted as part of this study and are included in Chapter 5.

The same mobile unit used at Hamilton Harbor was tested at the Middlefield Landfill in Bay City, MI, for another field-scale demonstration. Oils, water, and soils were treated in this demonstration for USEPA and Environment Canada. PCBs and trichloroethylene (TCE) were the target contaminants. A 72-hr continuous run was planned at process rates of between 3 and 15 tons/day.[1]

Efficiency

Results from a completed field test indicate that PCB-destruction efficiency is between 99.9998 and 99.9999 percent. This was based on the results of one test run of Hamilton Harbor sediments spiked with 500 ppm of PCB-containing oils (Campbell and Hallet 1992). PCB testing was limited because the contaminants of concern for this test were coal tar and PAHs. Results indicated that the process is very efficient for organic contaminants. Stack emissions were far below allowable levels; the scrubber decant water was acceptable at a local municipal wastewater treatment facility; and the scrubber sludge was a nonhazardous waste suitable for disposal at a landfill. The reactor solids had a very low organic content but were contaminated with metals (Campbell and Hallett 1992).

Dioxins and furans also were not contaminants of concern and so were not analyzed in the influent. However, the stack emissions were analyzed for furans to determine if they were being created in the reactor. Concentrations less than 0.39 mg/m^3 were recorded (Eco Logic 1992a).

Pretreatment requirements

If the sediment is to undergo thermal desorption, only screening to 5 mm and mixing or blending of the material is required. However, if the TDU is not used and the untreated sediment is introduced directly to the thermo-chemical reduction process, the suspended solids concentration of the feed must be adjusted to between 8 and 20 percent. If hydraulic dredging was possible, material handling would be reduced, and the optimum solids concentration of the slurry could be achieved during the dredging.

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[1] Personal Communication, 17 November 1992, J. Nash, ELI Eco Logic International, Inc., Rockwood, Ontario, Canada.

Pumping the slurry directly to the treatment site would be efficient, assuming location, distance, and vessel traffic would permit a pipeline to be used. Alternatively, dredged material could be transformed into a slurry during the unloading process from the barge. A temporary controlled storage or confinement site would be necessary because the rate of dredging would exceed the processing capacity of a full-scale unit capable of 100 tons/day. Leachate control mechanisms would be required. The leachate could be returned to the input waste stream and processed.

Particles greater than 5 mm in diameter are too large for processing through the reactor vessel and are required to be screened out of the waste stream. Once separated, the particles are washed with slurry water and again with clean water. At the Hamilton Harbor demonstration site, this washed gravel portion has been found to be clean enough under the RCRA standards to be accepted as a nonregisterable landfill material. The wash water can be returned to the slurry preparation tank for treatment.

Posttreatment requirements

Air emissions, scrubber decant water, and scrubber sludge make up the total effluent. Sampling of air emissions from the boiler stack, which burns a mixture of propane or natural gas and 5 percent of the scrubbed gases, indicates that contaminant concentrations during the Hamilton Harbor field testing were well below local ambient air quality standards and no further treatment was necessary. However, monitoring of the stack emissions would continue.

In the Hamilton Harbor demonstration, the scrubber decant water was sufficiently free of contaminants to be accepted for conventional sewage treatment. Insufficient data is available as yet on the testing of dioxin-bearing sediments to determine the level of posttreatment, if any, for the decant water.

The scrubber sludge was more contaminated in the same demonstration. PAHs concentrations as high as 12 ppm were recorded. However, the sludge was considered sufficiently clean for placing in a landfill. The volume of sludge produced during the demonstration was approximately 1 percent of the volume of the sediment treated (Eco Logic 1992a).

Overall, the process appears capable of destroying target contaminants to such an extent that all effluents, while containing traces of contaminants, can be classified as nontoxic and can be disposed of without further treatment. Monitoring of the three effluents will be required to ensure that appropriate contaminant levels are not exceeded. Any of the effluents can be recirculated into the process stream for further treatment, if required. However, testing of dioxin and PCB contaminants has not been sufficient to determine if effluent control and posttreatment of effluents will be necessary for sediments containing these compounds.

Feasibility of full-scale implementation

Processing equipment for this treatment can be delivered and operated on two 50-ft trailers. Two other smaller trailers contain the process control equipment and the monitoring equipment. According to the developer, a full-scale unit with a capacity of 100 ton/day (85 yd^3/day) can be built in the same configuration and can be installed in a space of 65 by 200 ft.

Pretreatment transportation needs and storage volumes are dependent upon the amount of contaminated sediment that is to be processed. The dredging and transport of the sediment to the treatment site can be accomplished at rates in excess of the processing rate of this treatment. Therefore, the storage volume required is the same as the volume of sediment to be dredged.

Based on the Hamilton Harbor field test, effluent volumes will be 90 percent of the input water. For a 20-percent solids mixture, 21,500 gal of water per day will be input, and 19,300 gal/day of liquid effluent will be created. This effluent is reported to be acceptable for conventional sewage treatment. The volume of sludge created is estimated to be approximately 1 percent of the liquid effluent or 215 gal/day, based on the full-scale production rate.

Posttreatment transportation requirements will be concerned primarily with the disposal of the treated sediments and are not a function of the treatment process. Disposal of the scrubber sludge will be required.

Assuming a full-scale processor capable of 100 yd^3/day, with a conservative downtime of 20 percent, 10,000 yd^3 of sediment could be treated within 5 months, and 50,000 yd^3 in a little under 2 years, using a single processing unit. Multiple units would decrease the overall treatment time in proportion to the number of units employed. Two units theoretically could treat 50,000 yd^3 in 13 months and 100,000 yd^3 in 26 months.

Limiting maximum concentrations of contaminants have yet to be defined. Experience to date shows that up to 30 percent dry-weight of PAHs and 500 ppm of PCBs can be treated successfully. Concentrations of up to 40 percent PCBs in oil (diluted to 20 percent in the influent) and 1,000 ppm in both soil and water are planned to be treated at the Bay City, MI, field test that is now in preparation. Conceptually, no waste concentration would be too great, provided sufficient dilution can be achieved in the influent stream. No minimum concentration limits are reported in terms of efficiency of treatment.

Because the influent stream is required to be greater than 80 percent water, this process is not sensitive to sediment characteristics except that particle size is limited to 5 mm in diameter or less. This is a function of the physical design of the equipment.

Based on the Hamilton Harbor field-testing experience, establishing the treatment plant at the contaminated site is most economical, reducing material handling. The process, even at commercial size (100 tons/day) can be transported on four trailers. A level graded area of 65 by 200 ft is required. A water supply, electricity, and sewer connection are necessary, although, as at Hamilton Harbor, a diesel generator can supply sufficient electricity if necessary. The plant could be barge-mounted if required, assuming water and sewer connections were available.

Besides the integral processing, control, analytical, and monitoring equipment, tanks of propane, hydrogen, and nitrogen are required. Earthmoving equipment, capable of handling 100 tons a day is necessary to deliver the untreated material to the slurrying tank and to remove the treated material from the reactor discharge area.

Human health and environmental impacts related to the process do not appear to be significantly greater than for normal industrial processes. Effluent and emission analyses during the field tests have indicated that ambient air and water quality standards were met at the field test locations (Ecologic 1992a).

Public concerns raised by the neighboring community will reflect the location of the treatment site. For example, at Hamilton Harbor, the surrounding area was zoned for light industrial uses. The Bay City site is close to residential areas and adjoins a park. At community meetings held before processing began at Hamilton Harbor, several concerns were raised. These included questions on emissions monitoring, the ownership of hazardous material, the explosive nature of hydrogen used in the process, the possible future land use of the treatment site, spills and emergency response capabilities, and many others related more to the existence of the contaminated material than to the treatment process. Because this process is essentially transportable, no permanent installation is necessary; thus public acceptance may be more easily achieved than in the case of a permanent processing plant and its associated importation of contaminated material.

Costs

The costs estimates (Table 8) for a full-scale commercial treatment system for the Eco Logic proprietary thermal gas-phase reduction process, with a feed rate of 100 yd^3/day, are based on two different projects, Hamilton Harbor, Ontario, and Bay City, MI, where the process has been used to treat PCB-contaminated sediments and soils. The costs are provided by Eco Logic.[1]

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[1] Personal Communication, January 1993, Kelvin Campbell, ELI Eco Logic International, Inc., Rockwood, Ontario, Canada.

Table 8
Cost Analysis: Thermal Gas-Phase Reduction Process

Item	Cost $/yd^3	Type of Cost	Present Value Cost $,000s for		
			10,000 yd^3	50,000 yd^3	100,000 yd^3
Site preparation		Fixed	100	100	100
Permitting and regulatory costs		Fixed	100	100	100
Capital equipment		Variable	247	1,005	1,837
Start-up and fixed costs		Fixed	100	100	100
Pretreatment of wastes		Variable[1]	0	0	0
Labor costs	35	Variable	346	1,675	3,214
Consumables, supplies, and utilities	260	Variable	2,573	12,443	23,879
Effluent treatment and disposal	5	Variable	49	239	459
Monitoring and analytical costs[2]	20	Variable	197	957	1,837
Maintenance and repair costs	5	Variable	49	239	459
Site demobilization and cleanup		Fixed	98	92	84
Dredging	7.5	Variable	75	375	750
Transportation to TSF[3]	15	Variable	150	750	1,500
Construction of TSF[4]		Variable	300	1,000	1,800
Land lease for TSF[4]		Variable	5	52	247
Land lease for process equipment[4]		Variable	3	13	25
Disposal of residual material		Variable	0	0	0
Landfill disposal	67.28	Variable	666	3,220	6,179
Total, $,000s			5,062	22,360	42,571
Cost, $/yd^3			506	447	426
Throughput (feed rate) in yd^3/day			100	100	100
Period for completion (in months) (using 5 working days per week)			4.6	23.1	46.2
Monthly discount rate (assuming 4.4 percent annual discount rate)	0.37%				

[1] These costs are implicit in other costs.

[2] These costs include royalties of $10/yd^3.

[3] These costs are estimated at $30/yd^3 for 10,000 yd^3, $20/yd^3 for 50,000 yd^3, $18/yd^3 for 100,000 yd^3 and 1 million yd^3.

[4] Land lease costs are based on an annual lease rate of $7,000/acre.

Site preparation. These costs will be partially site-specific and partially dependent upon the land area required for the treatment equipment, exclusive of the contaminated sediment storage area. This system is fully transportable and can be set up in a relatively small area with minimal preparation. This category is not expected to add a significant amount to the overall treatment costs. For comparison purposes, it is estimated the site preparation costs are about $100,000.

Permitting and regulatory requirements. Special site considerations and the type of waste to be treated can strongly influence these costs. The type of treatment technology can also influence permitting procedures; those processes that destroy contaminants by conversion to nontoxic substances and that do not use hazardous materials in the treatment process are more likely to obtain permits with less effort. However, because the target contaminants are dioxins and PCBs, regulatory requirements and permitting may be a significant factor in terms of time and money. These costs are estimated to be $100,000.

Capital equipment. The unit capital cost of a 100-yd^3/day unit is estimated at $25, $21, and $20 per cubic yard of processed material.

Start-up and fixed costs. Start-up includes integrity testing and shakedown, but does not include transportation to the site or assembly of the processing equipment. Fixed costs are defined as the costs required to operate the processing unit and are independent of the waste throughput. Start-up and fixed costs are estimated to be $100,000.

Pretreatment of waste. Pretreatment costs are included in the treatment process.

Labor costs. The assumption was made that labor requirements would require two engineers, four trained operators, eight laborers, one maintenance person, and one administrator for the operation of a full-scale unit for a total of $3,500 per day.

Consumables, supplies, and utilities. In this case, electricity generation and influent process water were included. Variable costs, dependent upon waste throughput, are estimated to be directly proportional to the processing rate. The costs are estimated to be $260 per cubic yard of material processed.

Effluent treatment and disposal. Because the effluent, the decant water, has been shown to be nontoxic and suitable for disposal into the municipal wastewater system, these costs are low and estimated to be $5 per cubic yard.

Disposal costs. Based on the two demonstration projects, all residual materials were suitable for disposal in a sanitary landfill, along with the treated sediment. Using the assumptions described in Chapter 2, the total transport and disposal costs are estimated to be $67 per cubic yard.

Monitoring and analytical costs. These costs have been estimated to be $20 per cubic yard. These estimates from the vendor are based on their experience with PCB and PAH remediations. It is likely that monitoring and analytical costs for dioxin-contaminated sediments may be significantly greater, especially when dealing with part-per-billion concentrations.

Maintenance and repair costs. These costs are estimated at $5 per cubic yard.

Site demobilization. For the process equipment and associated trailers and gas storage tanks, the total demobilization and cleanup costs were estimated to be $100,000.

Solvent Extraction

Solvent extraction is potentially effective in treating oily sludges and saturated soils contaminated with PCBs, PAHs, dioxins, and pesticides by separating the sludges into three fractions: oil, water, and solids. As the fractions separate, contaminants are partitioned into specific phases. For example, PCBs are concentrated in the oil fraction, while metals are separated into the solids fraction. The overall volume and toxicity of the original waste solids are thereby reduced, and the separate concentrated waste streams can be treated more efficiently for disposal. A treatment process using solvent extraction that shows a promising application for remediating contaminated sediments is the Basic Extractive Sludge Treatment (B.E.S.T.) system.

Description of treatment process

The B.E.S.T. process is a mobile solvent extraction system that uses one or more secondary or tertiary amines (usually triethylamine (TEA)) to separate organics from soils, sediments, and sludges. This process has been patented by the Resources Conservation Company (RCC), Ellicot City, MD. This technology is based on the solubility characteristics of TEA, which is completely soluble in water at temperatures below 20 °C. This principle is exploited by using the oil and water components of the sediment in conjunction with the TEA to create a single-phase extraction medium, which is a homogeneous mixture at near ambient temperatures and pressures. This allows the B.E.S.T. process to handle feed mixtures with relatively high water content without reducing extraction efficiency (Tose 1987).

The extraction process begins by mixing and agitating cold solvent and pH-adjusted waste in a washer/dryer. The washer/dryer can be a horizontal steam-jacketed vessel with rotating paddles. Hydrocarbons and water in the waste simultaneously solvate with the cold TEA, creating a homogeneous mixture. As the solvent breaks the oil-water-solid emulsions in the waste, the solids are released and allowed to settle by gravity. The solvent mixture is decanted, and fine particles are removed by centrifuging. The resulting dry solids have been cleansed of hydrocarbons but contain most of the original heavy metals.

The solvent mixture from the washer/dryer unit (containing the organics and water extracted from the waste) is heated. As the temperature of the solvent increases, the water separates from the organics and from the solvent. The organics-solvent fraction is decanted and pumped to a stripping column where the solvent is recycled, and the organics are discharged for recycling or disposal. The water phase is passed to a second stripping column where residual solvent is recovered for recycling. Typically, the water is discharged to a local wastewater treatment plant after monitoring and analysis. Depending upon the level and type of contamination, this water fraction may be returned to the marine environment without further treatment, depending on the trace amounts of oil and sediments remaining. Figure 7 shows the process flowchart. Figure 8 shows a B.E.S.T. solvent cleanup unit schematic.

The B.E.S.T. technology is modular and expandable, allowing for onsite treatment. Based on the results of many bench-scale treatability tests and some full-scale remedial actions, the process significantly reduces the hydrocarbon concentration in the solids. By removing contaminants, the process reduces the overall toxicity of the solids and water streams. It also concentrates the contaminants into a smaller volume, allowing for efficient final treatment and disposal.

The B.E.S.T. process can be used to remove most organic contaminants from sludges or soils, including PCBs, PAHs, pesticides, furans, and dioxins. However, performance can be influenced by the presence of detergents and emulsifiers, low pH, and reactivity of the organics with the solvent.

State of development

The process has been evaluated under USEPA's SITE Demonstration Program and found to be effective in treating oily sludges and soils contaminated with PCBs, PAHs, and pesticides. Full-scale commercial units utilizing the B.E.S.T. process have been constructed.

A full-scale soils treatment unit was used at the General Refining Superfund site in Garden City, GA. The unit processed up to 70 tons per unit of PCB- and metals-contaminated sludges per day. This USEPA emergency response action cleaned 3,700 tons of sludges during an operational period of 7 months. PCBs were detected at low concentrations (5 to 15 ppm) in the oily sludges prior to treatment and to <0.1 ppm levels in the solids after treatment (USEPA 1990a).

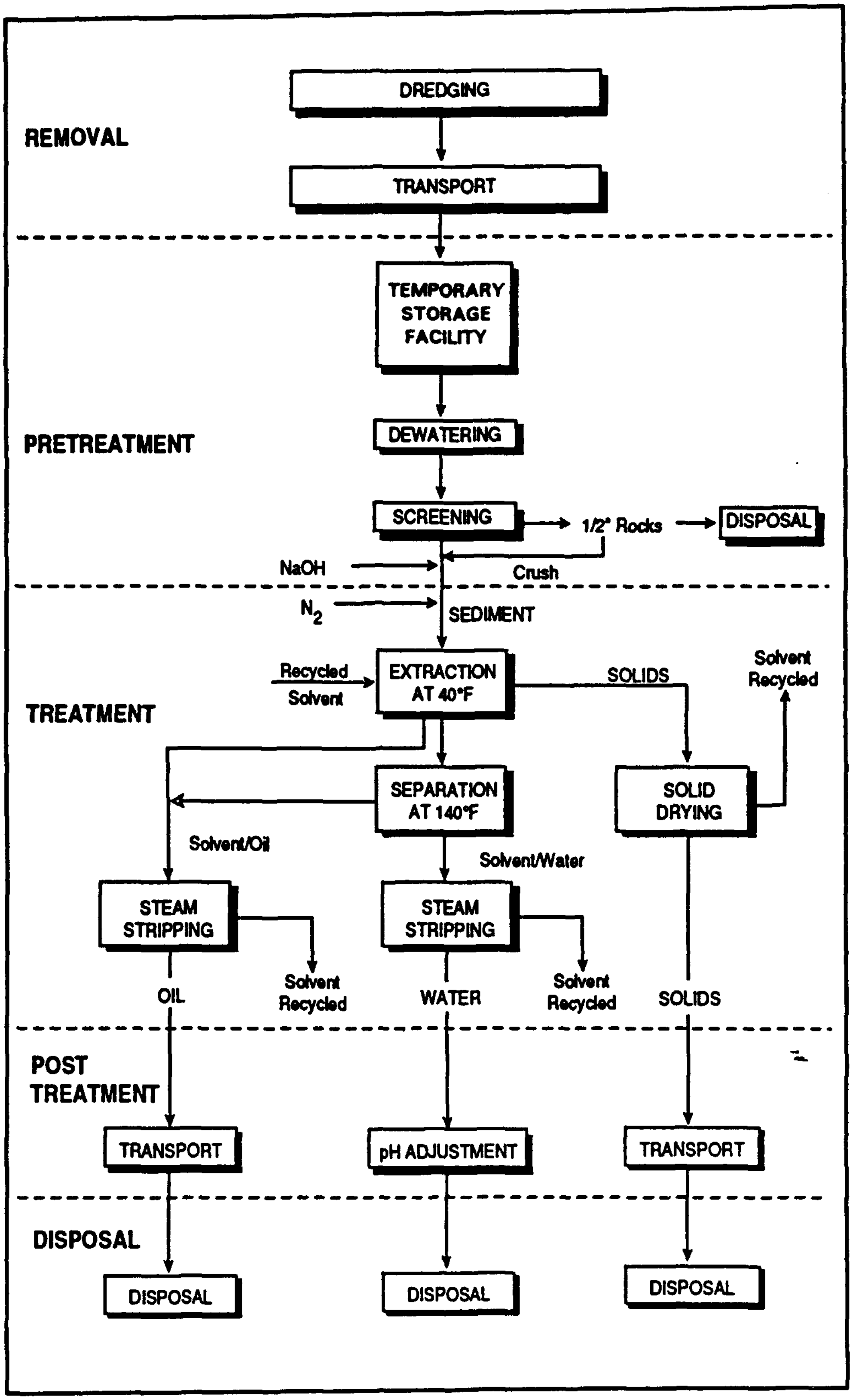

Figure 7. B.E.S.T. process flowchart

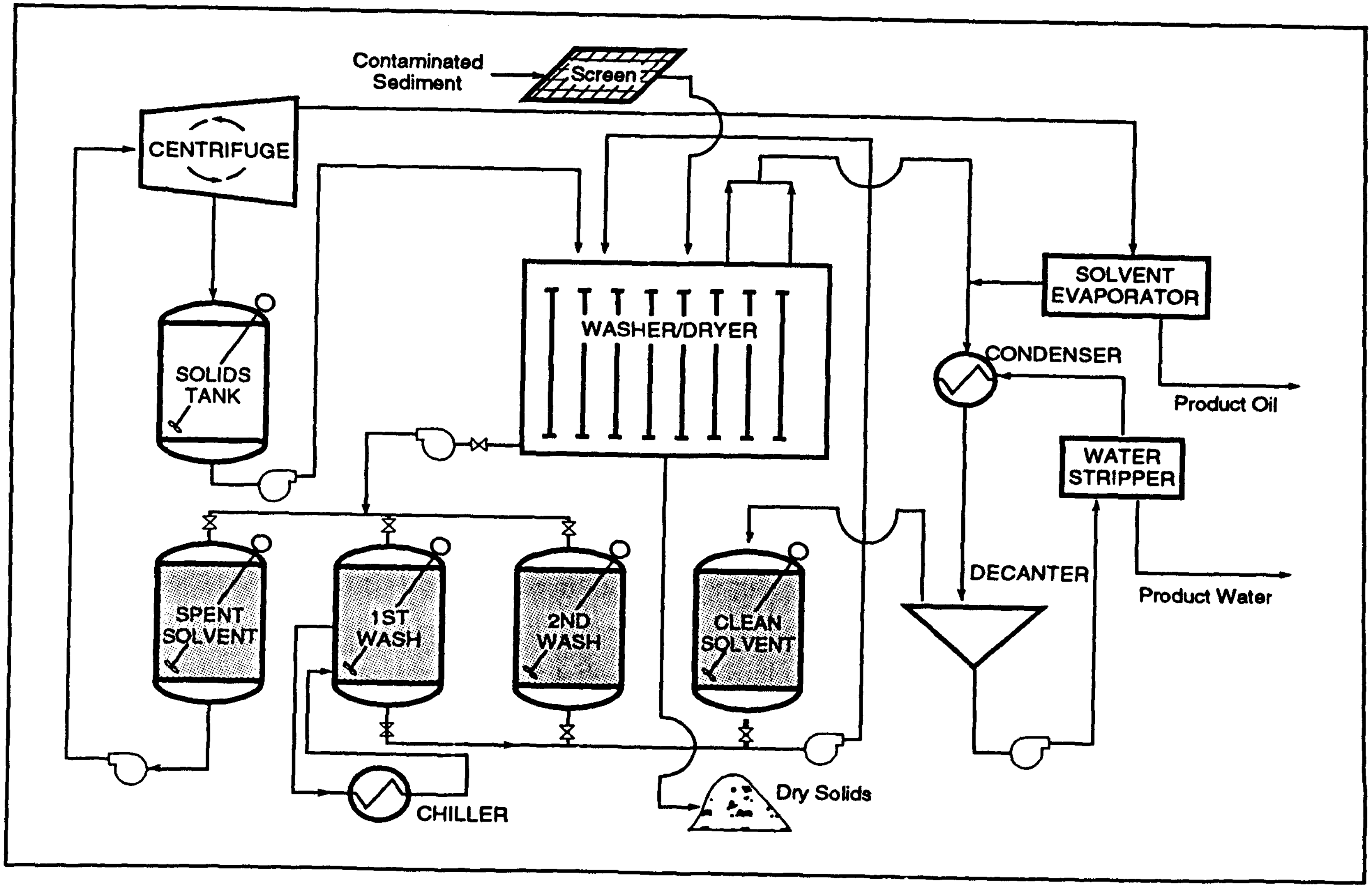

Figure 8. B.E.S.T. solvent extraction process schematic

A pilot-scale unit is available. The unit is capable of treating 100 lb per day of dry solids and has a nationwide TSCA permit for onsite demonstration testing with PCB-contaminated wastes. The unit was used as a demonstration at a site in the midwestern United States for removing PCBs from soil material. Approximately 1,000 lb of soil with a PCB level of 130 ppm was treated. Cleanup levels near 2 ppm were obtained, with no PCBs detected in the recovered water (detection limit of 20 ppb).[1]

Solvent extraction is the selected remedial action at the Pinnete's Salvage Yard site in Maine, the Ewan Property site in New Jersey, the Norwood PCB site in Massachusetts, and the Alcoa site in Massena, New York. Also, it is the preferred alternative at the F. O'Connor site in Maine. The demonstration of the B.E.S.T. process under the SITE Demonstration Program was performed in summer 1992 at Indiana Harbor, with the final report due in summer 1993. Sediment with a PCB concentration of 12 ppm was treated with a greater than 99-percent removal efficiency (USEPA 1992b).

The removal efficiency of toxicity equivalency of dioxin exceeded 99 percent for both high-level and low-level PCB-containing untreated soil. The majority of pilot studies were targeted for PCB contamination of sludges and soils, as shown in Table 9. Dioxin removal data is unavailable from any of these pilot plant studies. A summary of available bench-scale testing performed for PCB- and dioxin-contaminated soil is included in Table 10.

Table 9
Bench-Scale Testing Results for Dioxin in PCB-Contaminated Soils

	High PCB		Low PCB	
	Untreated Soil	Treated Soil	Unsaturated Soil	Treated Soil
Oil and Gas, percent	1.6	<0.2	<0.2	<0.2
Toxicity equivalency	7.845	0.06329	0.366	0.000075

Efficiency

Solvent extraction is potentially effective in treating oily sludges and materials. The process extracts oil and grease from contaminated material. PCBs, PAHs, furans, and other organic compounds that are solubilized in the oil and grease are also removed in the process.

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[1] Personal Communication, January 1993, L. Weimer, Resources Conservation Company, Bellevue, WA.

Table 10
State of Development B.E.S.T. Solvent Extraction Process

Test Date	Matrix Tested	Target Contaminant(s)	Source of Contamination	Initial Concentration	Treated Solids Concentration	Test Objective
Jul 1992	Sediment	PCBs, PAHs	Primary Steel, Coke, Petroleum	PCBs-650 mg/kg PAHs-85,560 mg/kg	PCBs-1.0 mg/kg PAHs-670 mg/kg	PAH Removal >96%
Sep - Oct 1991	Sludge	PCBs	Aluminum Manufacturing	530 mg/kg	0.7 mg/kg	PCBs <2.0 mg/kg
Sep - Oct 1991	Soil	PCBs	Aluminum Manufacturing	800 mg/kg	1.0 mg/kg	PCBs <2.0 mg/kg
Sep - Oct 1991	Sludge	PCBs	Aluminum Manufacturing	480 mg/kg	1.0 mg/kg	PCBs <2.0 mg/kg
Sep - Oct 1991	Sludge	PCBs	Aluminum Manufacturing	137 mg/kg	0.6 mg/kg	PCBs <2.0 mg/kg
Sep - Oct 1991	Sludge	PCBs	Aluminum Manufacturing	13 mg/kg	0.3 mg/kg	PCBs <2.0 mg/kg
Sep - Oct 1991	Soil	PCBs	Aluminum Manufacturing	5 mg/kg	0.2 mg/kg	PCBs <2.0 mg/kg
Jun 1991	Soil	PAHs	Wood Treatment Wastes	10,900 mg/kg	109 mg/kg	Determine BDAT Std.
Jun 1991	Soil	PAHs	Wood Treatment Wastes	14,000 mg/kg	8.2 mg/kg	Determine BDAT Std.
Dec 1989	Soil	PCBs	Machining Ops. Lubricant Disposal	130 mg/kg	2.5 mg/kg	PCBs <10 mg/kg
Feb 1989	Sludge	Oil and gas, PAHs	Petroleum Refining	Oil and gas - 26%	Oil and gas-0.09% PAHs-11.6 mg/kg	K048-K052 BDAT Std.

Note: Reference—Resource Conservation Company (1993).

The removal efficiency for organic compounds directly depends on the removal efficiency for oil and grease. If oil and grease is a significant portion of the waste, the removal efficiency can exceed 99 percent in a single pass or extraction stage. The removal efficiency drops to 98 to 99 percent when oil and grease levels drop to ppm levels. Subsequent passes or extractions can result in higher removal efficiencies and can exceed 99 percent for materials containing ppm levels of contaminants. Removal efficiencies decrease with decreasing contaminant levels

The removal efficiency for PCBs and dioxins will depend on the initial concentrations of these compounds in the waste, initial concentrations of oil and grease in the waste, and solubility of PCBs and dioxins in the oil and grease. This process has achieved 98 to 99 percent removal efficiency for PCBs, depending on the nature of the wastes treated. Removal efficiencies for dioxins have been evaluated to a very limited extent.

However, dioxins are expected to show the same characteristics as PCBs, and removal efficiencies for dioxins should range from 98 to 99 percent.

The B.E.S.T. process extracts the organic contaminants from oily soils and sludges and separates the waste into three distinct streams: solids, oil, and water. The organic contaminants are concentrated in the oil stream; the solids and water streams contain trace amounts. The oil stream must be further treated before disposal.

Pretreatment requirements

The B.E.S.T. method has no specific requirement either for removal of sediments from the harbor or for transportation of the sediments to the treatment plant or holding/dewatering area. However, because the process operates more efficiently at low moisture contents, clamshell dredging may be beneficial in decreasing the need to separate free water before treatment.

Pretreatment also includes screening the contaminated feed solids to remove cobbles and debris for smooth flow through the process. Screening of particles larger than one-half inch is necessary. Screening has been incorporated into the process. The particles refused by the screen may be broken or crushed and reintroduced.

Dewatering of the sediments may be beneficial for saving costs. The weight of materials being introduced into the treatment plant will be reduced by dewatering. The treatment plant itself has no limitation on water content. Whether dewatering is an option or not will depend heavily on two factors:

- Chemical concentrations in the effluent water.
- Costs of installing, operating, and maintaining a dewatering unit or facility.

At a minimum, an engineered TSF should be employed in the pretreatment phases of the sediment. TSFs serve several functions such as the following:

- Providing temporary storage of dredged material.
- Allowing for settling and dewatering of solids.
- Separating coarse-grained or oversized material from fine-grained sediments.

A TSF also allows for the settling of suspended sediments to achieve acceptable concentrations for discharging into receiving waters.

Because TEA is flammable in the presence of oxygen, the treatment system must be sealed from the atmosphere and operated under a nitrogen blanket. Prior to treatment, it is necessary to raise the pH of the waste material to greater than 10, creating an environment where TEA will be conserved effectively for recycling through the process. This pH adjustment may be accomplished by adding sodium hydroxide prior to mixing the sediment with TEA (Tose 1987).

Posttreatment requirements

The process effluents are as follows:

- Solids—1 to 5 percent moisture, trace quantities of target organic compounds.
- Wastewater—pH of 10 or higher, trace quantities of target organic compounds.
- Oil—up to 2 percent wastewater, 1 to 2 percent fines, target organic contaminated compounds.

If necessary, the solids will require further treatment to remove inorganic contaminants such as heavy metals, which will require disposal at an appropriate solid waste disposal facility. The choice of the disposal facility will depend on the concentrations of hazardous materials in the treated solids. Open-water disposal of solids may be allowed if applicable criteria are met. The choice of the disposal facility for wastewater will depend on the concentrations of hazardous materials in the wastewater. The pH may have to be adjusted before disposal. The oil containing residual contaminants would have to be recycled or incinerated. The choice of posttreatment method will depend on the concentrations and types of organic compounds in the effluent. The solvent is recycled into the process. If soluble metals are present, precipitation or other removal processes must be applied before recycling.

Runoff. The surface runoff from a TSF or the treatment plant can be controlled by properly grading the surface. To prevent offsite migration of contaminated sediments, the runoff can be captured in detention or settling basins. The captured solids can be fed into the treatment plant.

Leachate. There is no possibility of leachate being generated by this process. Seepage of water from a TSF may be controlled by installing a liner and subsurface drainage structure. Seepage water can be collected and treated if necessary.

Air emissions. The treatment unit does not release any emissions into the air. Triethylamine is a noxious gas and is an irritant. The treatment plant has several controls to prevent accidental releases into the atmosphere.

Feasibility of full-scale implementation

The treatment costs would range from \$100 to \$200 per cubic yard, depending upon the volume of material processed and water content. Costs of dredging, transportation and dewatering sediment, and posttreatment and disposal of effluents would be additional.

The effluent volumes can be calculated by a simple mass balance equation. The volume of soil, water, and oil generated will depend on influent conditions. Table 11 demonstrates on an order of magnitude basis a range of volumes, based on an average total recoverable petroleum hydrocarbons (TRPH) of 1,510 ppm and a total solids of 52 percent.

Table 11
Estimated Effluent Volumes

Contaminated sediments, yd^3	10,000	50,000	100,000
Solids, yd^3	5,200	26,000	52,000
Water, million gal	3.6	18	36
Recovered oil, thousand lb	20.3	101.4	202.8
Temporary storage facility, acres	2 to 3	5 to 7	14 to 16

The TSF size was conservatively estimated using a ponding depth of 2 ft and a storage depth of 11 ft. The actual design of the TSF will depend on several factors including volume. These factors include sediment characteristics, initial slurry concentration, results of column settling tests, effluent suspended sediment concentration requirements, and available area. A bulking factor of one is assumed for converting in situ volumes to dewatered volumes. The bulking factor and associated pretreatment volumes will vary depending on sediment type and dredging techniques.

The treatment plant will require approximately a 1-acre area. If needed, the treatment plant can also be installed on a barge. An engineered TSF will require substantially more space depending on dredged material volumes. However, barged dredged material could be scheduled to match the 216-yd^3/day processing rate.

It is estimated that it would take approximately 12 to 18 months to set up the treatment plant capable of processing 216 yd^3/day. This includes obtaining necessary permits prior to commencing treatment.

Time required for cleanup will depend on the volume of sediments processed and whether the sediments will be dewatered. Assuming a treatment volume equal to the dredge volume with a unit weight of 1.16 tons per cubic yard, treatment time would be 2, 11, and 22 months for volumes of 10,000, 50,000, and 100,000 yd^3, respectively, for a 250-ton/day

(216-yd^3/day) plant. This does not include plant setup time, but does assume 5 working days per week.

The efficiency of removal depends on the influent contaminant concentrations and characteristics and the number of extraction stages. The removal efficiency drops to 98 to 99 percent at ppm levels of contaminants.

Availability

The B.E.S.T. extraction process is a proprietary process. Several mobile treatment plants have been constructed and have performed full-scale commercial operations. The most economic approach to installing a processing unit onsite (either construction of a new unit or the installation of an existing mobile unit) is dependent on the quantity of material to be processed at that site.

The hours of operation will depend on local zoning laws and constraints on the project schedule. Typically, previous units have been operated on a 10-hr per day schedule.

The primary adverse human health impact during construction could be from accidental release of TEA into the atmosphere. TEA is a flammable, noxious gas. However, the olfactory threshold of the gas is very low, and adequate precautions are built into the process to prevent accidental releases.

Regulatory requirements

Implementation of the B.E.S.T. process would require review and permitting by several local, state, and Federal regulatory agencies. Regulatory requirements on a local level would include a building permit, a grading permit, and possibly a conditional use permit to accommodate local zoning laws. It is also possible that an environment impact assessment of the cleanup plant processes will be required.

On a state level, disposal of the posttreatment water resulting from the B.E.S.T. process may require permitting. If the effluent water is pH adjusted, it may be allowed to be placed back into the adjoining waterway; this would require a National Pollutant Discharge Elimination System (NPDES) permit as administered by the local water quality control board. The effluent water may be shipped or piped into a local wastewater treatment plant as an alternative. This option would require a local connection or disposal permit as well. Offsite transportation of the extracted oil fraction would need to be conducted by a licensed hazardous waste transporter per state and Federal regulaticns.

Costs

The cost estimates are shown in Table 12 for a full-scale commercial treatment for the B.E.S.T. solvent extraction process with a daily feed rate of 215 yd^3 of material. These costs are provided by the technology developer and are based on field tests for PAH- and PCB-contaminated sediments from the Grand Calumet River, IN. The demonstration was sponsored by USACE under the ARCS program, and USEPA SITE program. Present value adjustments have been applied.

Site preparation. These costs will be partially site specific and partially dependent upon the land area required for the treatment equipment, exclusive of the contaminated sediment storage area. This B.E.S.T. system is fully transportable and can be set up in a relatively small area with minimal preparation. This category is not expected to add a significant amount to the overall treatment costs. For comparison purposes, it is estimated that the site preparation costs are about $100,000.

Capital equipment. The capital cost of a unit with an effective processing rate of 216 yd^3/day is estimated at $12.70 per cubic yard of processed material for all three treatment volumes.

Start-up and fixed costs. Start-up includes integrity testing and shakedown, but does not include transportation to the site or assembly of the processing equipment. Fixed costs are defined as the costs required to operate the processing unit and are independent of the waste throughput. Start-up and fixed costs are estimated to be $400,000.

Pretreatment of waste. Pretreatment costs are estimated at $5 per cubic yard of material to be processed.

Labor costs. Unit labor costs are estimated to be $25.90 per cubic yard of material to be treated.

Consumables, supplies, and utilities. In this case, electricity generation and influent process water were included. Variable costs, dependent upon waste throughput are estimated to be directly proportional to the processing rate. The costs are estimated to be $52.40 per cubic yard of material processed.

Effluent treatment and disposal. This estimate was $1.40 per cubic yard.

Maintenance and repair costs. These costs are estimated at $3.60 per cubic yard of processed material.

Site demobilization. For the process equipment and associated trailers and materials handling equipment, the total demobilization and cleanup costs were estimated to be $300,000.

Table 12
Cost Analysis: Solvent Extraction Process

Item	Cost, $/yd^3	Type of Cost	Present Value Cost $,000s for		
			10,000 yd^3	50,000 yd^3	100,000 yd^3
Site preparation		Fixed	100	100	100
Permitting and regulatory costs		Fixed	100	100	100
Capital equipment	12.7	Variable	126	622	1,219
Start-up and fixed costs		Fixed	400	400	400
Pretreatment of wastes	5	Variable	50	250	500
Labor costs	25.9	Variable	258	1,268	2,487
Consumables, supplies, and utilities	52.4	Variable	521	2,565	5,031
Effluent treatment and disposal	1.4	Variable	14	69	134
Monitoring and analytical costs	6	Variable	60	294	576
Maintenance and repair costs	3.6	Variable	36	176	346
Site demobilization and cleanup		Fixed	298	285	277
Dredging	7.5	Variable	75	375	750
Transportation to TSF	15	Variable	150	750	1,500
Construction of TSF[1]		Variable	300	1,000	1,800
Land lease for TSF[2]		Variable	2	24	120
Land lease for process equipment[2]		Variable	1	6	12
Disposal of residual material[3]	600	Variable	6	30	58
Landfill disposal	67.28	Variable	669	3,293	6,460
Total, $,000s			3,166	11,609	21,871
Cost, $/yd^3			317	232	219
Throughput (feed rate) in yd^3/day			216	216	216
Period for completion (in months) (using 5 working days per week)			2.1	10.7	21.4
Monthly discount rate (assuming 4.4 percent annual discount rate	0.37%				

[1] These costs are estimated at $30/yd^3 for 10,000 yd^3, $20/yd^3 for 50,000 yd^3, $18/yd^3 for 100,000 yd^3.

[2] Land lease costs are based on an annual lease rate of $7,000/acre.

[3] Hazardous material disposal costs estimated at $200/ton in a RCRA landfill plus transportation costs. Residual oil is 2 ton/10,000 yd^3 of treated sediment.

Disposal of residual material. Disposal costs of the concentrated residual oil are estimated from an assumed 2,000-mile round trip to a permitted RCRA landfill at the rates described in Chapter 2 and a disposal cost of $200/ton of material.

Base-Catalyzed Dechlorination

This patented chemical dehalogenation process, developed at the USEPA RREL, has been demonstrated to destroy halogenated organic contaminants such as chlorinated solvents, PCBs, dioxins, and furans in soils, sediments, sludges, and other media (Rogers, Komel, and Sparks 1991). The principal treatment is based on the use of hydrogen as a nucleophile to replace halogen ions associated with the halogenated contaminant molecule.

This process is a refinement of methods that use alkaline metal hydroxides in polyethylene glycol (PEG), which are capable of partially dechlorinating compounds (U.S. Congress 1991). Steric hinderance of the relatively large glycol molecules is thought to be the reason why complete dehalogenation is not attainable using these reagents. However, hydrogen ions are much smaller and are capable of completely replacing the halogen molecules in the target compound. The conceptual process reaction is shown in Figure 9. Hydrogen radicals, generated from the hydrogen donor R′ in the presence of a base (e.g., NaOH), and a catalyst at temperatures between 250 to 350 °C replace the chlorine ions in $R\text{-}Cl_x$ to produce $R\text{-}H_x$, a dechlorinated compound, and a chloride salt (NaCl). The loss of hydrogen ions produces R″ from R′ (ETG Environmental 1992).

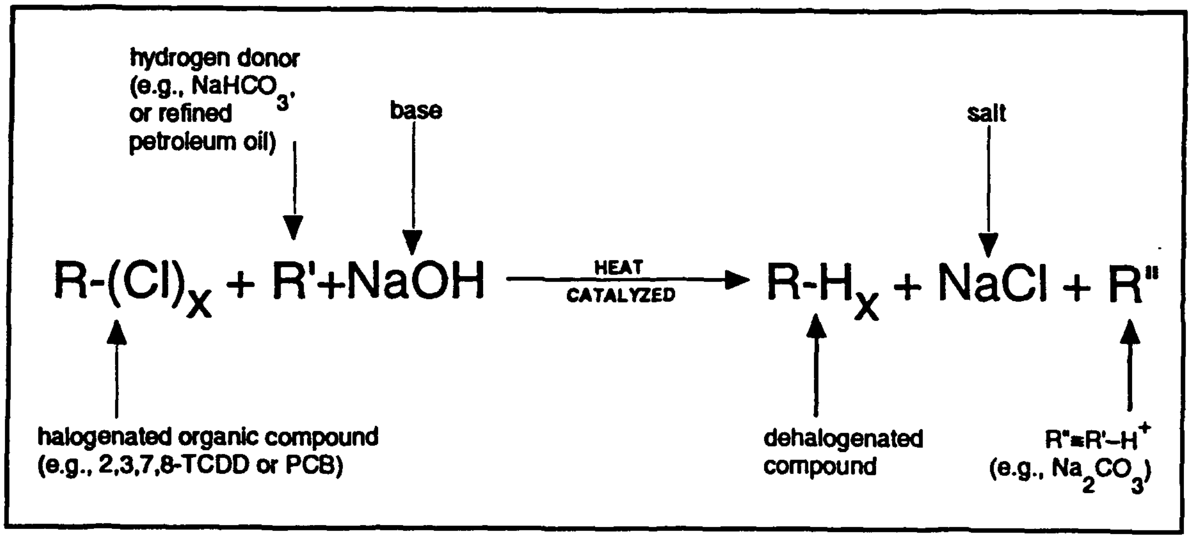

Figure 9. Conceptual base-catalyzed dechlorination reactions

Description of treatment process

Figure 10 is a flowchart of the required remediation components describing the overall treatment procedure from removal to disposal. As with nearly all sediment treatment technologies, dredging and transport to a treatment site is necessary. After treatment, the clean material requires final disposal, as does the effluent and residuals produced as a result of the process.

Sodium bicarbonate is mixed with the material (10 percent by weight) as a dechlorination reagent before being conveyed to the rotary reactor, where the solid-phase reaction takes place. The second treatment step is the liquid-phase reaction, carried out in a stirred-tank reactor. The feed material is slurried and mixed with the base and catalyst. Following dechlorination, the slurry is separated; process water is filtered and can be discharged to a wastewater treatment plant for further treatment and disposal. The solid fraction can be disposed of as a nonhazardous waste (Chan and Yeh 1992).

A detailed process schematic of the treatment is presented in Figure 11. Sediment and catalyst are heated to 350 °C (630 °F) for 1 hr as the material passes through the rotary reactor. Treated material is discharged to soil hoppers for cooling, and samples are analyzed to verify the extent of contaminant destruction. Reactor offgases, containing sediment and dust particles and trace quantities of volatilized organics are directed through a cyclone/baghouse, a venturi scrubber, and activated carbon filters. The cyclone/baghouse filters particulates from the gas. Offgases are then wet scrubbed and condensed with water in the venturi scrubber. The condensate is cooled and recycled through a settling tank to remove suspended solids. Scrubber water is treated by the addition of activated carbon, and the resultant slurry is dewatered in a filter press. The filtrate is again treated in an activated carbon filter and collected in a storage tank. The filter cake, particulates collected from the cyclone/baghouse, and the spent carbon filter material are processed further in the pressurized liquid-phase stirred-tank reactor (Chan and Yeh 1992).

A high-boiling point hydrocarbon oil, sodium hydroxide, and a proprietary catalyst are added. The oil is added to remove any remaining hydrophilic contaminants from the water. Nitrogen gas is injected into the reactor to displace oxygen and reduce the potential for explosive conditions being created in the tank. The combined-process wastes from the solid-phase reactor are mixed with the reagents to create a slurry that is agitated and heated to 350 °C for 2 hr in the liquid-phase reactor. Offgases from this step are condensed and filtered through activated carbon filters. The spent carbon is recycled into the reactor. After the liquid-phase reaction is complete, the decontaminated sediment is transferred to a sludge tank for cooling, sampling, and disposal once the contaminant concentrations of the treated material have been determined (Chan and Yeh 1992).

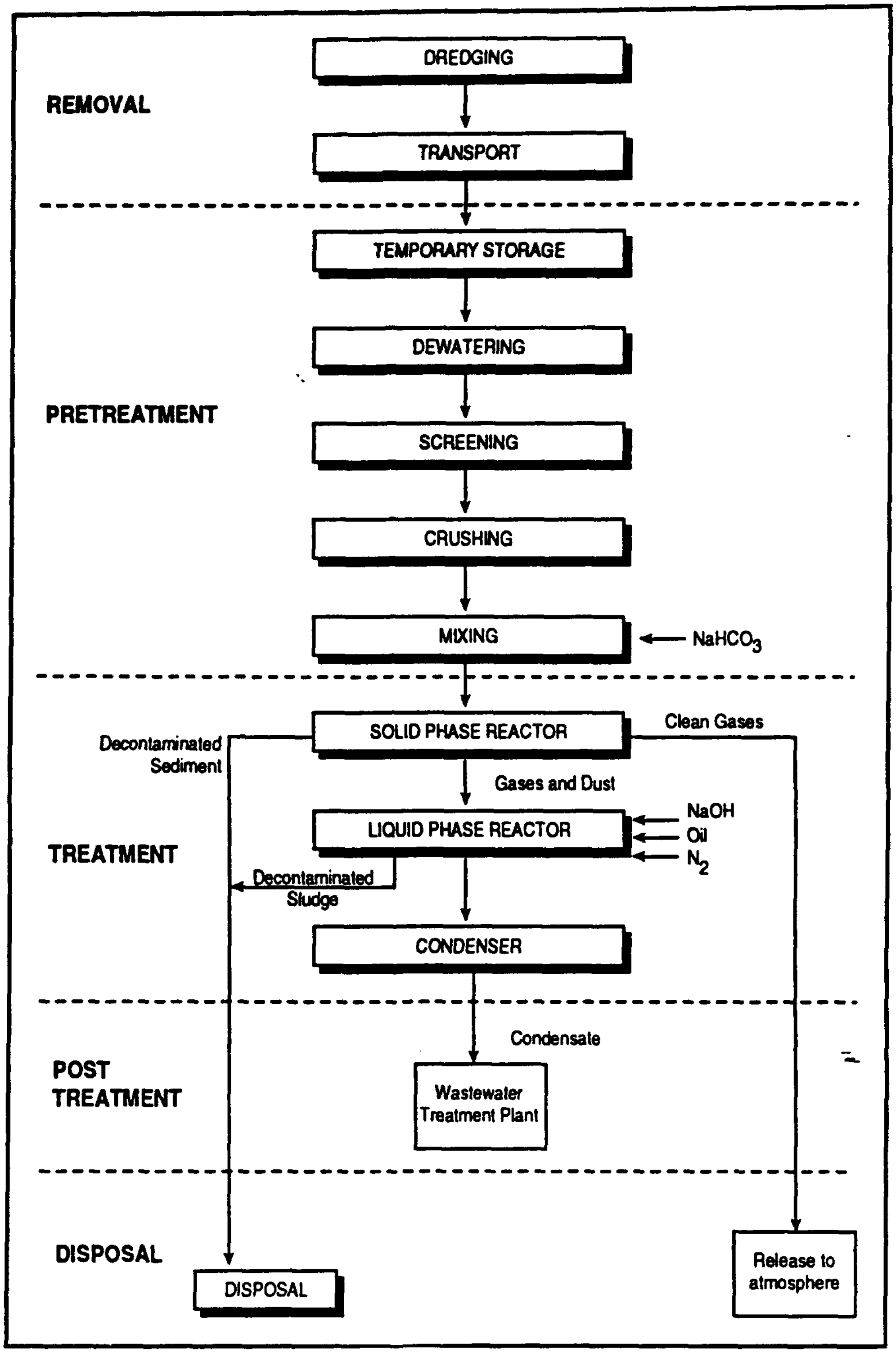

Figure 10. Base-catalyzed dechlorination process flowchart

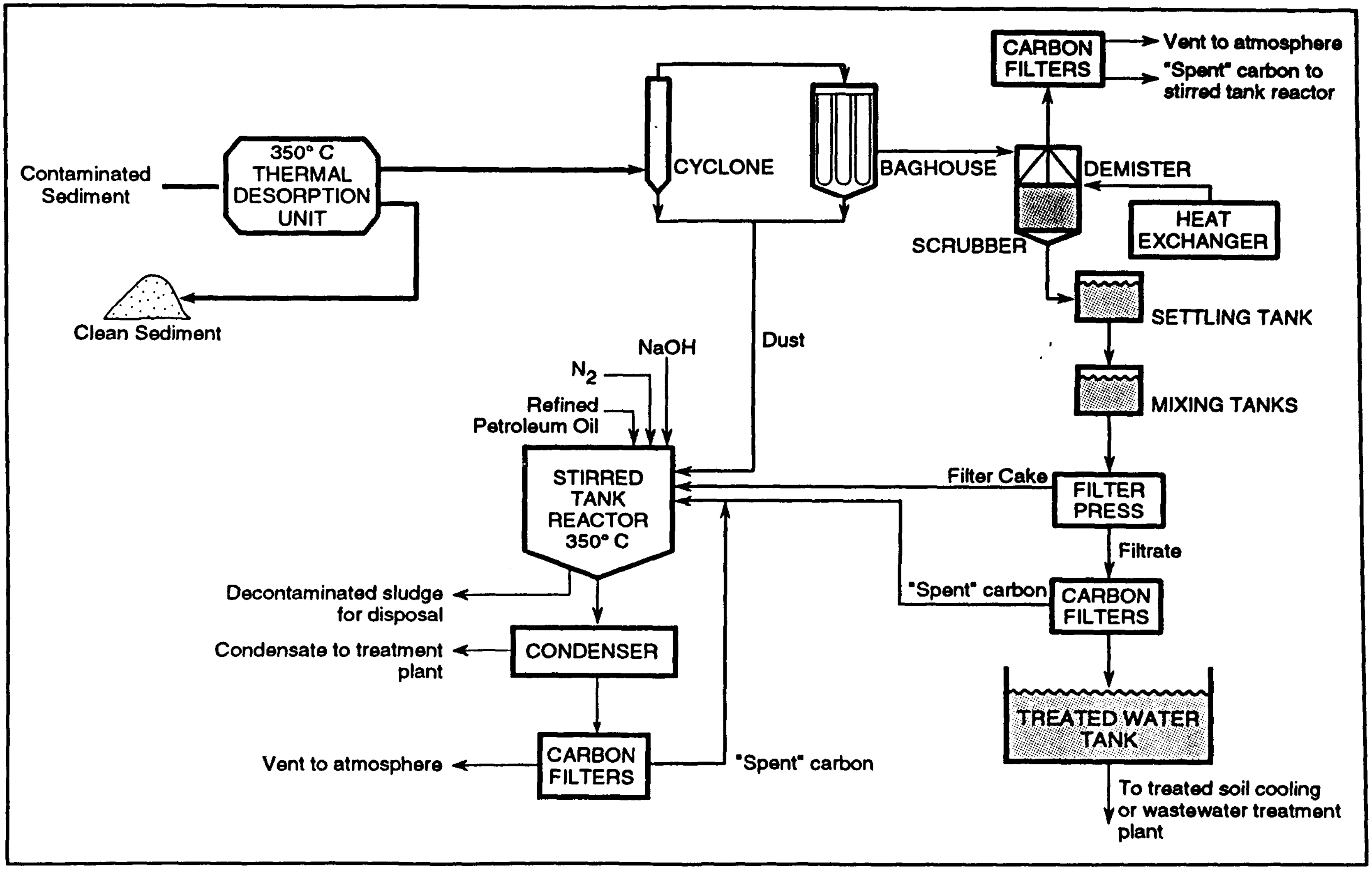

Figure 11. Base-catalyzed decomposition process schematic

State of development

Only a few field-scale tests of the BCD process of PCB-contaminated materials have been completed. Other tests are about to be performed (Table 13). These include a Superfund site at Brant, NY, treatment of sediment from Waukegan Harbor, IL, and soil at a U.S. Navy station on Guam (Rogers 1992). To date, only laboratory-scale testing of dioxin-contaminated soils has been attempted. Destruction efficiencies of greater than 99.99 percent were achieved from one test, performed by Wright State University (Tiernan 1992). Another test, for USEPA Region VII, is in progress. A third test is being carried out in conjunction with this study by Wright State, using sediment samples from New York Harbor (Chapter 5).

Table 13
State of Development: Base-Catalyzed Dechlorination Process

Developer/ Sponsor	Scale of Tests	Material Tested	Volumes or Processing Rates	Target Contaminants	Destruction Efficiency %	Comments	References
RREL/ USEPA-SITE	Field	Soil	49,000 cu yd	PCBs (100 to 6,000 ppm)	Not available	Planned for Superfund Site, Brant, NY	1
RREL USEPA	Field	Sediment	10 tons/hr	PCBs	99.9999	Waukegan Harbor, IL	1
RREL/U.S. Dept. of Energy	Bench	Liquid wastes	22 L reactor	PCP (with dioxins) (79 to 95%)	>99.9999	Idaho Falls, ID	5
RREL/U.S. Navy	Field	Soil	5,000 tons 1 ton/hr	PCBs (25 to 6,500 ppm)	Not available	In preparation at PWC Guam	4
RREL/ USEPA Region 7	Bench	Liquid/solid pesticide waste	—	Dioxins	Not available	In progress	2
RREL/ Wright State University	Bench	Soil	—	Dioxins/Furans	>99.99	Report in preparation	3
RREL/ USACE	Bench	Sediments	—	Dioxins	Not available	In progress	

Note:
1 = Rogers (1992).
2 = USEPA (1992c).
3 = Tiernan (1992).
4 = NEESA/NCEL (1991).
5 = Personal Communication, 1 December 1992, C. Rogers, USEPA RREL, Cincinnati, OH.

Efficiency

Results from one completed field test indicate that PCB destruction efficiency is between 99.9998 and 99.9999 percent. This was based on the results of a test run of Waukegan Harbor sediments (Rogers 1992). Other field tests are presently in progress or have been completed recently, and results have yet to be published.

Dioxins and furans destruction has been reported in only one laboratory test effort at the present time. Seventeen different dioxin/furan congeners with untreated concentrations ranging from 0.2 to over 2,000 ppb were analyzed. Only the congener with the highest concentration, octochlorinated dibenzo-p-dioxin, at 2,061 ppb, was detected in the treated material. A destruction efficiency for this congener was calculated at 99.99 percent. No other congeners were detected after treatment. Using the detection limits of the analyses minimum efficiencies of greater than 99.5 percent were calculated for the nondetected dioxin/furans (Tiernan 1992).

Pretreatment requirements

The process is capable of treating material with any range of water content. However, a moisture content of 10 to 15 percent is optimal in terms of volume reduction in the first treatment phase, which requires heating the material. The process also can work with dry feedstock, although reagent mixing is more difficult. For high contaminant concentrations and fine clay materials, the addition of water to slurry the material increases the mixing efficiency. In general, typical dredged sediment moisture content will be greater than the treatment optimum, and dewatering probably will be beneficial.[1] Screening and sizing of the material is also required. Particles larger than 1 in. are required to be screened and crushed and may be remixed with the smaller particles or disposed of separately and without treatment if contaminant levels are low enough to satisfy regulatory requirements.

Posttreatment requirements

The treated material is expected to be completely free of chlorinated organics and thus is nonhazardous and does not require disposal at a registered landfill. The only residuals produced are biphenyls and olefins, which have low toxicity and solubility and sodium chloride. The dechlorinated oily residues, which contain dust, sludge, and activated carbon, can be recycled as a fuel supplement for industrial use or be treated and reclaimed. Both the treated waste and the condensate from the treatment process can be discharged to a wastewater treatment plant after being pumped through an activated carbon filter. The decontaminated sludge

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[1] Personal Communication, 1 December 1992, C. Rogers, USEPA RREL, Cincinnati, OH.

can be disposed of in the same way as municipal sewage sludge. Monitoring and analysis of these residuals are required to determine the efficiency of the entire treatment and to ensure that the residuals conform with regulatory requirements for disposal (NEESA/NCEL 1991).

Feasibility of full-scale implementation

Pretreatment transportation needs and storage volumes are dependent upon the amount of contaminated sediment that is to be processed. The dredging and transport of the sediment to the treatment site can be accomplished at rates far in excess of the processing rate of this treatment. Therefore, the storage volume required is the same as the volume of sediment to be dredged.

Posttreatment transportation requirements will be concerned primarily with the disposal of the treated sediments. This will be a function of the treatment process to the extent of the contaminant destruction efficiency achieved and whether the treated sediments are considered clean or not. Treatability testing will be required to optimize the engineering design and to determine the expected efficiency of the overall process. Disposal of the decontaminated sludge will be required.

A continuous feed, full-scale, 1-ton/hour system has been fabricated and is in the process of being mobilized at Guam. Design has been completed for a larger 20-ton/hour unit. The system uses standard, off-the-shelf equipment; the reactors are available from applications in the food and mining industries. Two private companies, SoilTech ATP Systems, Inc. (Englewood, CO), and ETG Environmental, Inc. (Blue Bell, PA), have developed and tested equipment that can serve as the reactor vessels for the process. Systems capable of processing 5 and 10 tons/hour are now available. A new, complete system can be delivered and assembled within 8 to 10 months (NEESA/NCEL 1991).

The complete 1-ton/hour BCD equipment is presently housed on six trailers, and this is expected to be reduced to four trailers with subsequent units. This includes the following:

- Screening and crushing equipment.
- A pug mill.
- The solid-phase rotary reactor.
- Cyclone, baghouse, scrubber, heat exchanger, settling and mixing tanks, filter presses, and carbon filters.
- The liquid-phase stirred-tank reactor, condenser, and carbon filters.

- Analytical laboratory equipment, including a gas chromatography/ mass spectrometry.

Onsite, a fully assembled unit requires about 1 acre of land on which to operate. The unit requires either diesel or natural gas for a fuel source and a 440-V electricity supply, along with a water supply and access to a wastewater treatment facility. Electrical and radiant heat treatment units with capacities from 5 to 25 tons/hour have also been designed (NEESA/NCEL 1991).

Skilled and trained personnel requirements for a typical process are expected to consist of a supervisory engineer, a process control operator, and analytical support for product and residue/effluent sampling and analyses. Material rehandling and feed control, as well as pretreatment screening, crushing, and mixing, will require additional personnel.

The dechlorination process does not appear to be limited by high concentrations of contaminants, providing the appropriate amounts and types of reagents are added.[1] That is, for PCB-contaminated material, 5 to 10 percent by weight of bicarbonate is required for the solid-phase reactor step, and a 2:1 ratio of sodium hydroxide (NaOH) to PCB concentration is required in the liquid-phase test. It is reportedly possible to treat PCB concentrations of 6,000 to 7,000 ppm in soil and sediments. Similar dioxin concentrations are expected to be treatable, too.

Holding times in each reactor and recycling material through the reactors can facilitate dechlorination of heavily contaminated material. If heavy metals are present, they can be converted to insoluble carbonates or hydroxides and will remain with the dechlorinated material, which may then require further treatment to stabilize these metal salts.

Regulatory requirements will vary depending upon the site or extent of contamination of the material to be treated. Two Research and Development Permits have been issued under TSCA, and RREL is proceeding with an application for national permitting, also under TSCA.[1] This permit will allow any TSCA-regulated wastes to be treated using the BCD process at any site (assuming compliance with local and state land-use zoning laws).

Costs

Because of the uncertainties resulting from lack of specific site and sediment contamination information, the vendor supplied a range of estimated fixed and variable costs.

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[1] Personal Communication, 1 December 1992, C. Rogers, USEPA RREL, Cincinnati, OH.

The average cost estimates are shown in Table 14 for a full-scale commercial system using the Soil Tech Anaerobic Thermal Processor (ATP) in conjunction with BCD for treatment of sludges with low concentrations of chlorinated organics, including PCBs. These minimum and maximum costs were provided by Soil Tech ATP Systems, Inc.[1]

Cost estimates supplied by the vendor were limited in detail. Fixed costs as supplied covered several of the categories itemized in other treatment cost analyses, including site preparation, permitting costs, regulatory requirements, and start-up. Demobilization and site cleanup costs were estimated at 20 percent of the fixed costs. Fixed costs for the use of the Soil Tech technology typically range from $1.2 million to $1.8 million for use of the 10-ton/hour (tph) ATP. This ATP would typically be used for projects requiring the treatment of less than 50,000 tons of material.

Thermal Treatment by Incineration

Description of treatment process

Thermal treatment involves the use of heat as the primary treatment agent. During thermal destruction or incineration, organic materials in the waste are reduced to carbon dioxide (CO_2) and water vapor (both of which exit through a stack). Other chemicals such as chlorine and phosphorus are captured by the pollution control equipment, whereas noncombustible materials such as heavy metals are retained in the ash. USEPA began to develop incineration technology for treating dioxin-containing materials after laboratory studies showed that dioxins broke down when exposed to temperatures in excess of 1,200 °C. Figure 12 shows the chemical transformation of dioxin from the incineration process. To test this process on a much larger scale, USEPA built a mobile research incinerator specifically designed to treat recalcitrant organic chemicals. The success of this research, in which the DRE of dioxin in treated waste exceeded

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[1] Personal Communication, January 1993, J. Hutton, Soil Tech ATP Systems, Inc., Englewood, CO.

2,3,7,8-TCDD

Figure 12. Dioxin incineration reaction

Table 14
Cost Analysis: Base-Catalyzed Dechlorination Process[1]

Item	Cost, $/yd^3	Type of Cost	Present Value Cost $,000s for		
			10,000 yd^3	50,000 yd^3	100,000 yd^3
Site preparation		Fixed[2]	0	0	0
Permitting and regulatory costs		Fixed[2]	0	0	0
Capital equipment		Variable	0	0	0
Start-up and fixed costs		Fixed[2]	1,200	3,000	3,000
Pretreatment of wastes	5	Variable	50	250	500
Labor costs		Variable	3,145	8,475	16,544
Consumables, supplies, and utilities		Variable[3]	0	0	0
Effluent treatment and disposal		Variable[3]	0	0	0
Monitoring and analytical costs		Variable[3]	0	0	0
Maintenance and repair costs		Variable[3]	0	0	0
Site demobilization and cleanup		Fixed[4]	293	714	680
Dredging	7.5	Variable	75	375	750
Transportation to TSF	15	Variable	150	750	1,500
Construction of TSF		Variable	300	1,000	1,800
Land lease for TSF		Variable	8	30	148
Land lease for process equipment		Variable	4	8	15
Disposal of residual material		Variable	0	0	0
Landfill disposal	67.28	Variable	663	3,277	6,397
Total, $,000s			5,888	17,879	31,335
Cost			$589	$358	$313
Period for completion (in months) (using 5 working days per week)			69 6.7	172 13.4	172 26.8
Monthly discount rate (assuming 4.4% annual discount rate)	0.37%				

[1] This process includes the Soil Tech ATP.
[2] These costs are included in the start-up and fixed costs.
[3] These costs are included in the labor costs.
[4] These costs are assumed to be 20 percent of total fixed costs.

99.9999 percent, led USEPA to adopt thermal treatment as the appropriate method for destroying dioxin-containing waste (U.S. Congress 1991). Figure 13 shows the conceptual process flowchart from removal through disposal.

A rotary kiln system (RKS) consists of a rotating primary combustion chamber, a gas-fired afterburner, and a primary air pollution control system consisting of a quench, venturi scrubber, and a packed-tower scrubber. An alternative primary air pollution control system consisting of a quench and an ionizing wet scrubber may also be used. Figure 14 shows a schematic drawing of the RKS. Table 15 summarizes the typical characteristics of a full-scale RKS.

Table 15
Rotary Kiln Operating Characteristics

	Full-Scale	
Characteristic	**State-of-Practice**	**State-of-Technology**
Heat input		
Mw	3 to 15	3 to 20
MMBtu/hr	10 to 50	10 to 70
Heat release		
kW/m^3	155 to 414	155 to 580
Btu/hr-ft^3	15,000 to 40,000	15,000 to 56,000
Kiln exit temperature		
°C	260 to 982	260 to 1,316
°F	500 to 1,800	500 to 2,400
Afterburner exit temperature		
°C	871 to 1,538	871 to 1,538
°F	1,600 to 2,800	1,600 to 2,800
Kiln residence time		
Solids, hr	Up to 2	Up to 2
Gases, sec	1 to 3	1 to 3
Afterburner residence time sec	1 to 4	1 to 4
Excess air		
Range, percent	75 to 210	75 to 210
Stoichiometric ratio	1.75 to 3.1	1.75 to 3.1
Natural gas or fuel oil		

Note: Reference—Acurex Corporation (1991).

The key component of the RKS is a refractory-lined slightly conical cylinder that rotates at a speed of 0.2 to 1.5 rpm. The rotation serves to mix the material and expose sufficient surface area to heat, and the slight conical shape promotes solids movement laterally. Residence time at 0.2 rpm is approximately 1 hr. Other components of the kiln incinerator includes a waste feed system, a secondary combustion chamber or afterburner, air pollution control equipment, and an emissions stack.

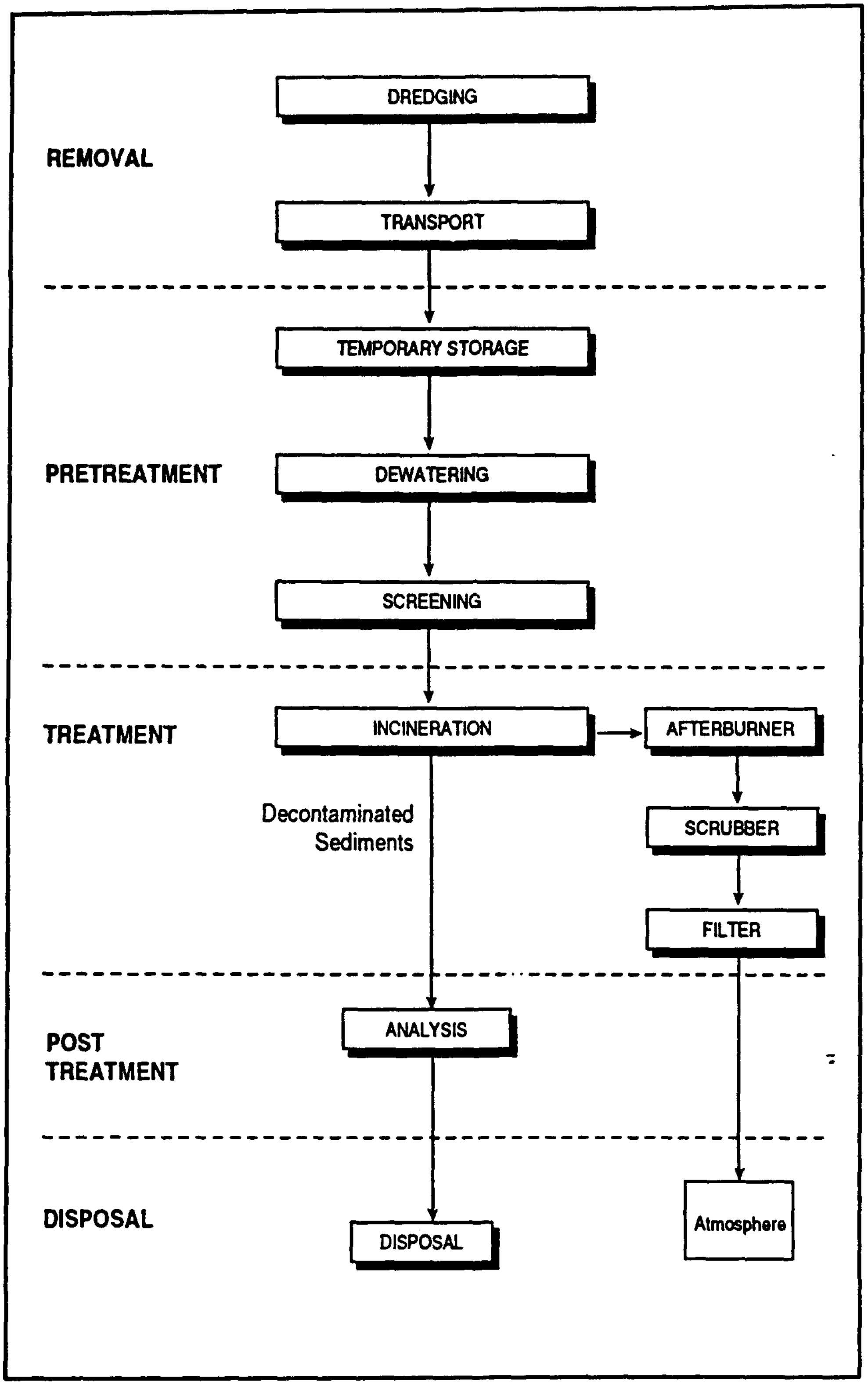

Figure 13. Incineration process flowchart

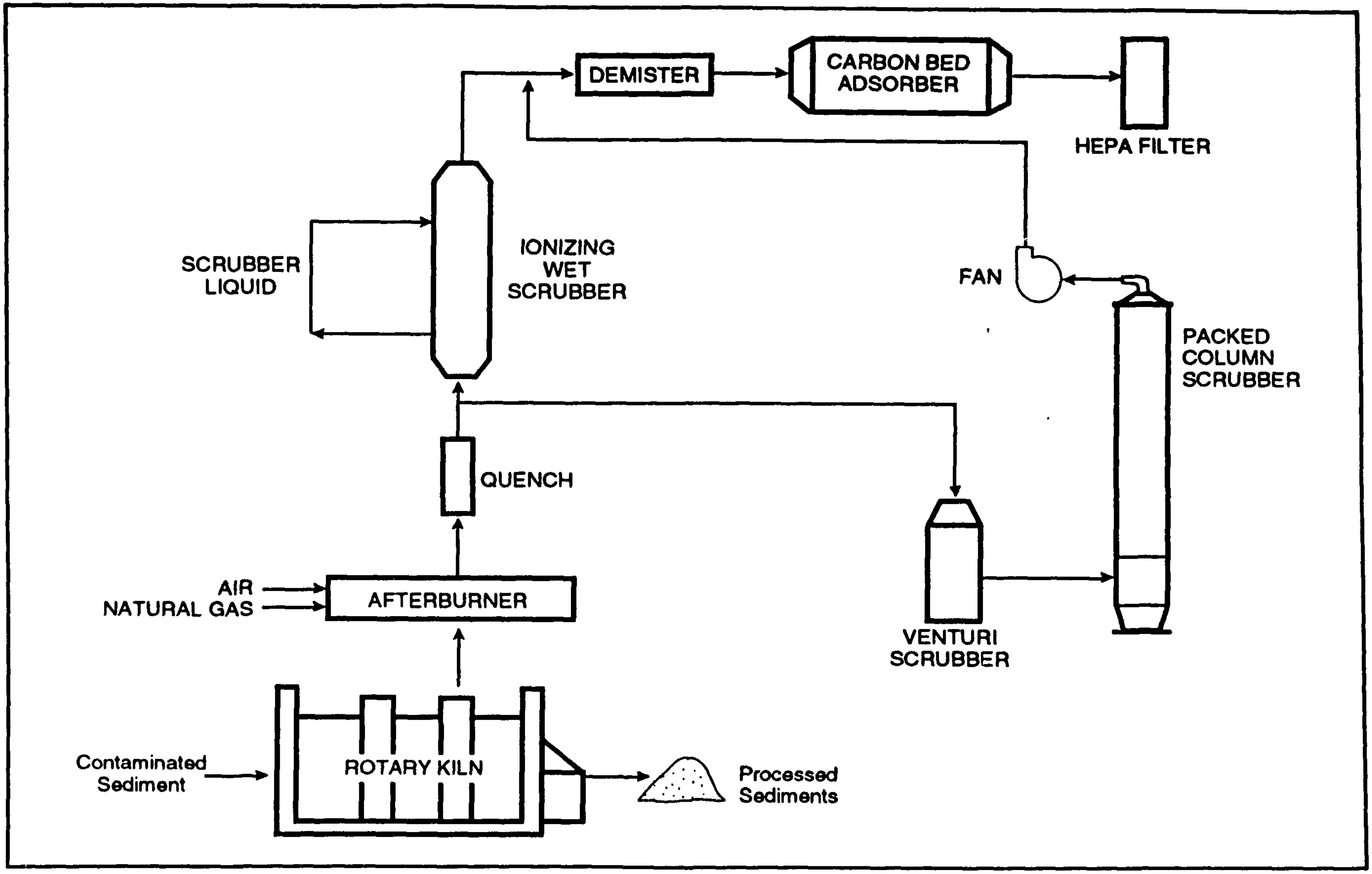

Figure 14. Rotary kiln incineration system process schematic

Solid and liquid wastes that are fed into the rotating kiln are partially burned into inorganic ash and gases. The ash is discarded in an ash bin, and the gaseous products in which noncombusted organic materials still reside are sent to the secondary combustion chamber for complete destruction.

State of development

At present, several commercial stationary rotary kiln facilities have permits to burn waste containing toxic constituents such as those permitted under the authority of TSCA. However, none of these facilities has been used to burn dioxins because of the lack of appropriate operating permits issued under RCRA and because of expected strong public opposition (U.S. Congress 1991).

Mobile rotary incinerators that can be transported and assembled at a field site have certain operating advantages over stationary facilities, the major one being that the potential risks and the permitting requirements for transporting hazardous wastes over long distances are avoided. USEPA, through the IRF, sponsored research on dioxin incineration technology beginning in 1985. The USEPA mobile incineration unit has successfully destroyed more than 12 million lb of dioxin-contaminated soils and 230,000 lb of dioxin-contaminated liquid waste between 1985 and 1986 (Gupta 1990). Only a few private companies have constructed incineration units based on the USEPA system. All of the systems are probably capable of destroying dioxin wastes, but again permitting and public concern considerations have inhibited application to dioxin-contaminated wastes (Table 16). The U.S. Air Force has demonstrated DREs greater than 99.9999 percent in tests of dioxin-contaminated soils from the Naval Construction Battalion Center in Gulfport, MS, using a mobile system based on the USEPA incinerator and developed by ENSCO Corporation, Little Rock, AR. Vesta, Inc. (Fort Lauderdale, FL), has developed an incineration unit that has been used successfully for the treatment of approximately 200 yd^3 of soil contaminated with dioxin-laden pesticides at Fort Hill, Bowling Green, VA, as well as other sites previous to 1987 (U.S. Congress 1991).

Although the USEPA mobile incinerator is no longer in use, the IRF has continued testing of incineration treatments. Bench-scale testing of sediments from the New Bedford Harbor superfund site achieved DREs greater than 99.9999 percent (in the flue gases only). For the tests, these sediments, containing PCB concentrations of almost 5,000 ppm, were spiked with PCBs to concentrations greater than 2 percent and incinerated without prior dewatering. The test sediment was spiked to a level that allowed an unambiguous determination of whether the regulatory DRE level could be achieved (Whitworth and Waterland 1992).

Table 16
State of Development: Incineration Processes

Developer/ Sponsor	Scale of Tests	Material Tested	Volume or Processing Rate	Target Contaminants	Efficiency %	Comments	References
IRF/USEPA	Field	Soil and liquid	12 million lb soil, 230,000 lb liquid wastes, 900 lb/hr solids, 180 gal/hr liquids	Dioxins	99.9999	USEPA Mobile Incineration Denney Farm, MO	2
ENSCO/U.S. Air Force	Field	Soils	2 to 5 yd^3/hr	Dioxins	99.9999	Naval CBC Gulfport, MS	1
Vesta/U.S. Army	Field	Soils	190 yd^3 4 yd^3/hr	Dioxins	99.9999	Fort Hill, Bowling Green, VA, and Others	1
Acurex/ USEPA	Bench	Sediments	100 lb	Dioxins	96.7	New York/ New Jersey Harbor Sediments	3
Acurex/ USEPA	Bench	Sediments	240 gal	PCBs	99.9999	New Bedford Harbor Superfund Site	4

Note:
1 = U.S. Congress (1991).
2 = Enviresponse, Inc. (1987).
3 = Acurex Corporation (1992).
4 = Whitworth and Waterland (1992).

Efficiency

Both earlier dioxin-treatment tests and more recent PCB-treatment tests indicate that incineration can successfully destroy these and other organic contaminants in soil and sediment matrices. DREs exceeding 99.9999 percent can be achieved in the stack emissions by adjusting waste feed rates, air and oxygen input, residence time, and incineration temperatures of both the kiln and afterburner; and it has been shown that dewatering of sediments is not necessary to achieve this level of destruction efficiency.

However, as shown by the New Bedford Harbor tests (treating sediment with an initial spiked PCB concentration of 46,000 ppm) with the solids resident in the kiln for 0.5 hr, the resultant ash (treated sediment) was still contaminated, ranging between 32 and 177 ppm of PCBs. This was equivalent to a solids decontamination effectiveness of 99.3 to 99.6 percent. The data collected indicated that incineration under the three test conditions was not sufficient to completely decontaminate the sediments. It is believed that the high moisture content of the feed material, an average of 64 percent, prevented the solids temperature from reaching levels needed for more complete PCB destruction. Increasing the residence time by a factor of two probably would be necessary (Whitworth and Waterland 1992). Because costs are proportional to kiln temperature and residence time, feed material volume reduction is desirable from an economic standpoint.

Pretreatment requirements

Prior to being fed into the kiln, oversized debris must be crushed or shredded into particles smaller than approximately 2 in. This creates a more uniform feedstock that is easier to handle; and it provides a larger surface area, increasing the efficiency of heat transfer, thereby decreasing the residence time and increasing the throughput. Dewatering the untreated sediment to approximately 20 percent also increases the efficiency of the incineration process by reducing the volume of material and thus the energy required to maintain incineration temperatures.

Posttreatment requirements

Incineration systems are composed of two separate steps: the first heats and decontaminates the solid fraction; the second treats the resultant gases. Assuming the solids are subject to the process at sufficiently high temperatures and for appropriate time periods, routine sampling and analysis will be required to show the material is decontaminated. The gas treatment and pollution control procedures are considered part of the treatment process and not a posttreatment. Again, routine monitoring of the stack gases will be a regulatory and operational requirement (to monitor the performance of the system). Effluent water from the gas treatment and cooling will require monitoring also. If contaminant levels exceed local wastewater treatment facility guidelines, the effluent could be recycled through the kiln, as can the scrubber filter cake used in the treatment.

Feasibility of full-scale implementation

Incineration systems, both stationary and mobile are currently in use for the treatment of contaminated wastes. Although no incinerator is permitted to burn dioxin-contaminated wastes at the present time, it is thought that many facilities are capable of achieving regulatory DRE levels if they were permitted (U.S. Congress 1991). Mobile incinerators will generally be of smaller capacity than stationary facilities and can also be delivered and installed onsite in less time than required to construct a stationary plant. For example, the USEPA mobile incinerator was contained on four trailers. Commercial mobile rotary kiln incinerator units are available from a number of developers. These units are reported to be capable of feed rates of up to 5 yd^3/hr and can be installed onsite in about 24 hr (U.S. Congress 1991). A stationary incineration treatment facility will require approximately 6 to 12 months to construct after regulatory approval has been granted.

Pretreatment transportation needs and storage volumes are dependent upon the amount of material to be processed. Dewatering is a beneficial pretreatment, increasing the efficiency of the incineration process by reducing the volume of material to be heated. If mechanical dewatering, crushing, or screening of the sediment are necessary, a larger site would be required to allow for the extra material handling. The mobile incinerator units can be assembled on approximately 1 acre; the dewatering, crushing, and screening will require 1 to 3 acres.

Posttreatment transportation of the treated and cleaned sediment will be required if the material is not to be stored permanently onsite. Air emissions are treated as part of the treatment process and should require no further processing. Effluent water and scrubber sludge will require recycling through the system or further treatment and disposal.

Limiting concentrations of contaminants (dioxins or PCBs) do not appear to be a problem with incineration technologies, especially at concentrations expected to be encountered in marine sediments. No minimum concentration limits are reported in terms of efficiency of treatment.

Commercially available mobile incinerators are reportedly capable of processing between 1.7 and 5.4 yd^3/hr of solid waste. This rate may decrease by half if sediments are processed without dewatering. Assuming 20 hr of operations per day, 10,000 yd^3 of dewatered sediment could be treated within 3 months, and 50,000 yd^3 in 5 months, using a single incinerator unit. Thirty months would be needed to treat 100,000 yd^3 of material. This time could be decreased proportionally by the number of incinerators used.

Incineration is capable of thermal destruction of all organic materials present in the feedstock. But, materials containing high levels of inorganic salts can cause degradation of the refractory materials in the kiln and slagging of the ash. High heavy metal content can result in elevated metal emissions, which are difficult to capture with air pollution control equipment (USEPA 1991b). Pretreatment processes may be required to address these circumstances. Heavy metal concentrations may also limit the ultimate disposal options for the incinerated solids.

The incineration process is hazardous because of the high temperatures maintained in the kiln and afterburner. Human health and environmental risks associated with the process are greater than for most other treatment processes because of the greater impact an emission control component failure could create. However, these hazards are proportional to the level of contamination of the waste; pretreatment dioxin levels in sediments are low, and the actual impact of a short-term process or equipment failure would be difficult to predict.

Construction or installation of a new onsite incineration facility would require compliance with all Federal and state regulations including RCRA and the Clean Air Act. RCRA design and operating permitting standards are listed under Subpart C, 40 CFR, Part 264. This permit would be administered by the state. Air pollution control permits controlling incinerator emissions would be issued by either state or local air quality control district agencies. A NPDES permit would be required also. Compliance with site-specific zoning and land-use regulations will be necessary also.

In the mid-1980s, the installation of the USEPA mobile incinerator at Denney Farm in Missouri met with community approval, and a request to extend operating permits for an extra year received no negative comments (Enviresponse 1987). Since then, public opposition to establishing incineration facilities in local communities has become widespread and has delayed and halted installations of municipal waste, as well as hazardous waste incinerators in several areas. One of the reasons for opposing the construction of the facility is the perception that once the incinerator is established, contaminated materials will be transported from other, nonlocal areas for treatment (U.S. Congress 1991). Another area of public concern over incineration is the creation of dioxins and furans as a by-product of incompletely combusted wastes that do not contain these compounds originally. Chronic effects on the health of the surrounding communities are a major concern also, as are concerns of negative environmental impacts from the combustion gases.

Costs

Commercial operators of incinerator facilities were unwilling to supply estimated treatment costs for contaminated sediments without more specific information on contaminant types and concentrations, sediment characteristics, and treatment locations. No cost figures on the treatment of dioxin-contaminated material was identified because no commercial incineration facility has been permitted to incinerate dioxins. However, because of the similar characteristics of PCBs and dioxins, it can be assumed that treatment costs will be similar, although permitting and regulatory requirements, as well as, effluent, emission, and residual monitoring and analytical costs, may be substantially greater for dioxins.

Total unit costs of $1,000 per cubic yard were estimated for all variable costs. It was assumed that a mobile incineration system would be used to treat the smallest volume (10,000 yd^3) of material and that a larger capacity incinerator would be designed and constructed onsite for the larger volumes. This cost was estimated at $5,000,000, and it includes site preparation, capital equipment, start-up, and other fixed costs.

Permitting efforts were estimated to cost $100,000. Again, this is very uncertain because of the lack of experience in incinerating dioxin wastes. Pretreatment costs include the construction and operation of a dewatering system. The analyses of estimated costs are shown in Table 17.

Table 17
Cost Analysis: Rotary Kiln Incineration Process

Item	Cost, \$/yd^3	Type of Cost	Present Value Cost \$,000s for		
			10,000 yd^3	50,000 yd^3	100,000 yd^3
Site preparation		Fixed[1]	0	0	0
Permitting and regulatory costs		Fixed	100	100	100
Capital equipment		Variable[1]	0	0	0
Start-up and fixed costs		Fixed[2]	500	5,000	5,000
Pretreatment of wastes		Variable	320	370	420
Labor costs		Variable	11,508	57,017	112,314
Consumables, supplies, and utilities	1,160	Variable[1]	0	0	0
Effluent treatment and disposal		Variable[1]	0	0	0
Monitoring and analytical costs		Variable[1]	0	0	0
Maintenance and repair costs		Variable[1]	0	0	0
Site demobilization and cleanup		Variable[1]	0	0	0
Dredging	7.5	Variable	75	375	750
Transportation to TSF	15	Variable	150	750	1,500
Construction of TSF		Variable	300	1,000	1,800
Land lease for TSF		Variable	4	19	94
Land lease for process equipment		Variable	2	5	9
Transport of residual wastes		Variable	0	0	0
Disposal in a landfill	67.28	Variable	667	3,307	6,514
Total, \$,000s		Variable	13,626	67,943	128,502
Cost			\$1,363	\$1,359	\$1,285
Throughput (feed rate) in yd^3/day			138	276	276
Period for completion (in months) (using 5 working days per week)			3.3	8.4	16.7
Monthly discount rate (assuming 4.4% annual discount rate)	0.37%				

1 These costs are implicit in other costs.
2 A mobile unit is rented for 10,000 yd^3, while a larger unit is constructed for larger volumes.

Bioremediation

Bioremediation processes are theoretically attractive for the treatment of dioxin-contaminated sites. However, current treatment processes have not progressed beyond bench-scale testing. A conference held at Rutgers University in May 1992 discussed bioremediation as a treatment technology for NY/NJ Harbor sediments and concluded that "A large scale bioremediation System is neither practical nor cost-effective at the moment. However, this situation should not preclude a demonstration project to investigate remediation methods relevant to New York Harbor sediments" (Institute of Marine and Coastal Sciences 1992). Although some research is underway for the remediation of contaminated soils and wastewater, very limited data has been compiled for dioxin, PCB, or for heavy metal contamination. (Chakrabarty, as cited in U.S. Congress 1991).

Because of the lack of proven technologies, either proprietary or generally available, no specific process can be assessed. Instead, a conceptual process illustrating a general approach to bioremediating dioxin-contaminated sediments is described. The treatment process could include contaminant degradation under aerobic conditions, anaerobic conditions, or a sequential combination of both.

Description of treatment process

This conceptual sequential anaerobic/aerobic treatment process is described as follows:

- Removal of contaminated sediment from the environment and its transport to a land-based treatment and storage facility.

- Storage for a period of months as a slurry, under anaerobic conditions for the initial dechlorination phase. Amendments, such as chemical agents, nutrients, or bacteria would be required.

- Preparation of a part of the sediment for aerobic dechlorination by debris removal, sizing, and soil washing of the coarse fraction.

- Transfer this portion to slurry bioreactors for aerobic bioremediation.

- Operation of the reactor with appropriate introduction of microorganisms, environmental conditions, nutrients, and adjustment of feed rates and residence times to provide for the maximum level of remediation.

- Repeat the aerobic process steps until all the sediment is treated to the desired level.

- Dewatering of slurry solids for disposal or reuse; or fill with capping; or reloading on barges for ocean disposal.

Figure 15 provides a flowchart of the process that combines unit operations into four general processes. The actual selection of individual steps within each process is dependent on the characteristics of contamination and the volume throughput required for a specific cleanup application.

A schematic diagram of tentative unit operations is shown in Figure 16. Unit processes have been selected for dealing with the sediment as a physical mixture of components. Unit processes have not been added that could be required to take into account other contaminants such as heavy metals or other organic constituents. Additional analytical information would be required to address the requirements for additional unit processes to treat these contaminants.

The following description of the unit processes is meant to be illustrative of the overall concept. It follows the treatment path of the "fine" sediment fraction and indicates points at which other fractions of the original sediment material are split off.

Bulk material would arrive at the storage and treatment facility and probably be removed from the barge by mechanical means or hydraulic pumping to a covered and diked storage area meant to eliminate exposure of the material to rain and contain any runoff or leachate so that it could be added to the process waters required in the bioreactor step. The sediment would be maintained under anaerobic conditions (i.e., ponded) in the storage area. Amendments to enhance the anaerobic dechlorination process would be added and the contaminant levels monitored regularly.

Removal of debris from the portions of sediment prepared for aerobic treatment is handled by mechanical sorting of the sediment using trommel/sifter/bar screen technology. The material continuing on from this step should be able to pass a 1/2-in. mesh screen (less than pebble-sized material). Debris will be collected and sent to the nearest appropriate landfill.

The next step, separation, consists of removal of the coarse fraction of materials, which have a high weight but a relatively small amount of surface area. Additional screening may be required with a final goal of removing the materials that are larger than 100 mesh. The viability of this step depends on the percentage of the sediment in the coarse fraction and the level of contamination of the coarse fraction.

The coarse materials will then proceed through a soil washing step in which an appropriate solvent/surfactant solution would be used to remove the organic coating on the particles. This coating contains the majority of the dioxin contaminant. Clean coarse material would be output and possibly be useful for fill material or other beneficial uses. Any beneficial use

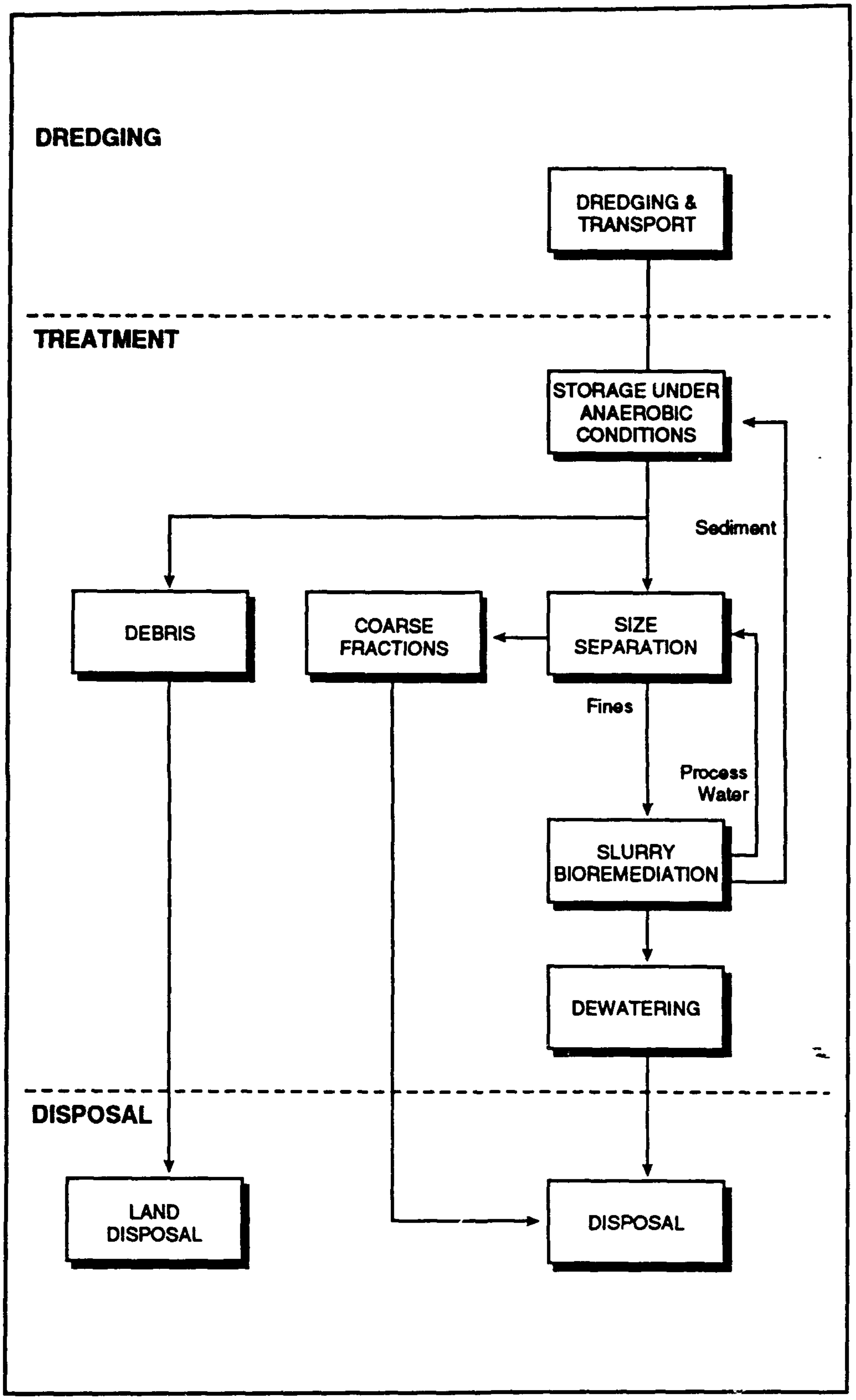

Figure 15. Bioremediation procedure flowchart

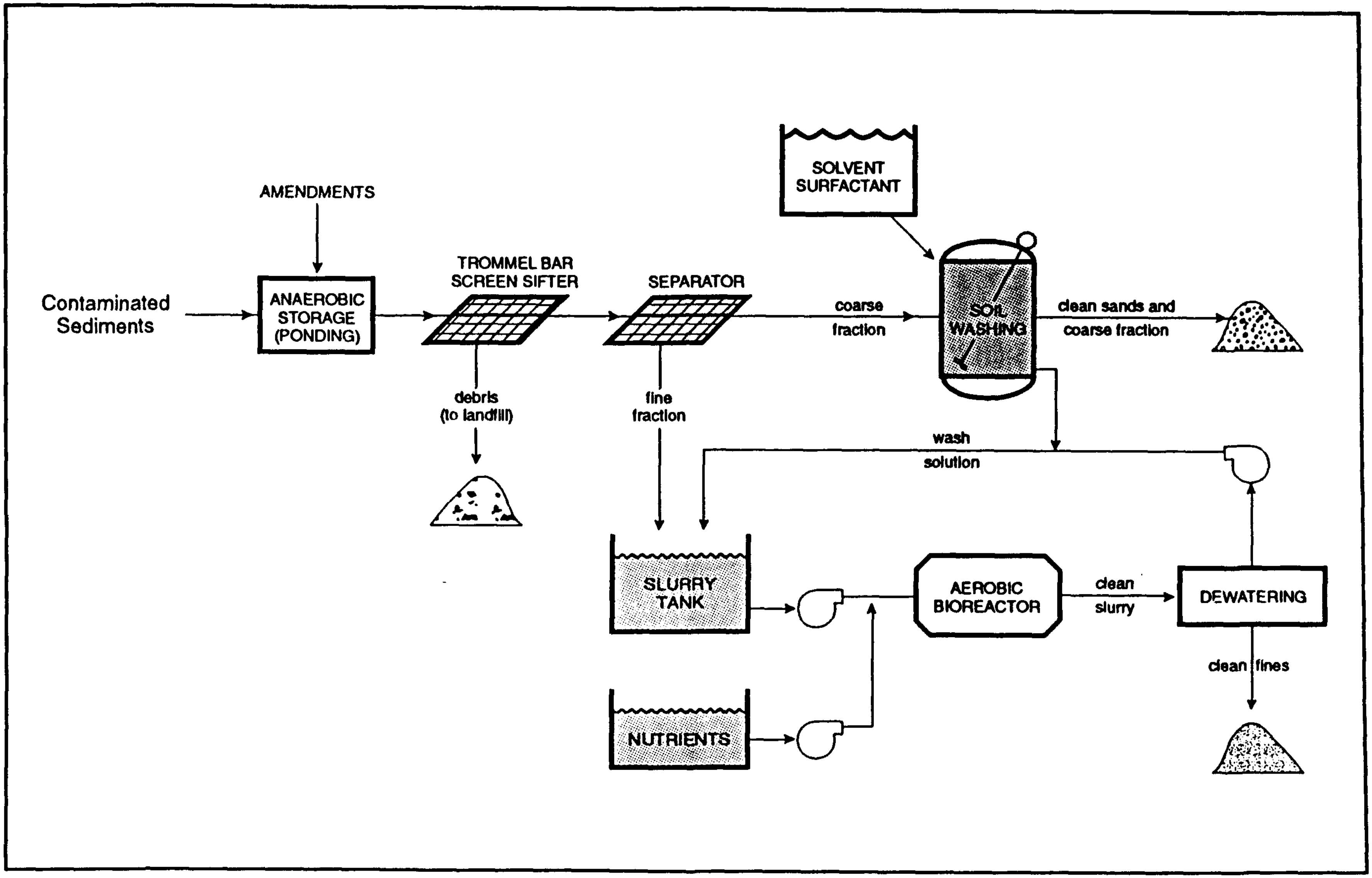

Figure 16. Bioremediation process schematic

would reduce the costs of final disposal, and if the material was of appropriate size, might be a saleable commodity.

The solvent solution carrying the contaminants will be routed to combine with the fine fraction that is output from the separation unit process, forming the slurry that will be treated in the bioreactor.

The bioreactor can be constructed with a volumetric capability up to and exceeding 100,000 gal. A combination of mechanical stirring and gas sparging provides the energy necessary to maintain a slurry suspension of 30- to 40-percent solids. At this solids loading, the effective viscosity of the working fluid is greatly increased, providing decreased settling velocities for the particles. Assuming a continuous flow process with a 30-day retention time for treatment completion, 35-percent solids in the slurry and 75 percent of the original sediment volume being fines of less than 63 μm diameter, the bioreactors will be required to process approximately 360,000 gal of slurry each month. Three to four reactors of 100,000-gal capacity will be required if stockpiling of untreated sediment is not possible. The bioreactors may be supplemented by an array of nutrient chemicals, pH adjustment, an oxygen source to maintain aerobic conditions, and a nitrogen source for maintaining the anaerobic conditions. Sampling and analysis capability will be required for process control monitoring and for confirming the decontamination of the processed materials.

Clean slurry will be withdrawn from the bioreactors and dewatered through a series of steps employing continuous flow centrifugation to recycle the water back to the slurry formation tank. Depending on the efficiency of the treatment, dewatered fines can be either returned for ocean disposal or possibly be used beneficially, such as for clean fill emplacement, or be returned to the storage area for permanent containment and capping.

State of development of treatment process

No commercial applications of this specific technological approach for the treatment of dioxin-contaminated material have been demonstrated or identified in the literature reviewed or during the interviews undertaken with government and private developers as part of this study. This conceptual process is not currently available as a treatment for dioxins in sediments because appropriate microbes and microbial pathways have yet to be identified for biotransformation of dioxins.

No data were found on field-scale tests either. A bench-scale test of PCB-contaminated sediments from New Bedford Harbor Superfund site in Massachusetts was performed in 1989 using aerobic bioremediation techniques with limited success. No bench testing of dioxin-contaminated sediments under anaerobic conditions have been identified. It has been suggested that the high acute toxicity, low solubility, and typically low concentrations of dioxins in contaminated soils and sediments have been a major hinderance to bioremediation research efforts (U.S. Congress 1991).

Corresponding to the lack of testing, accurate determinations of the efficiency of bioremediation processes are nonexistent. With respect to contaminated sediments, the data suggests that lower molecular weight and unhalogenated organic compounds will be preferentially reduced over more complex and halogenated organics. Even with these lighter, more volatile compounds, efficiencies are not high, being significantly less than 90 percent. Bioremediation process efficiency is affected by the presence of other organic components. Sediment characterization for a New York/New Jersey Harbor sample used for the treatability study discussed in Chapter 5 indicates an average total organic carbon content of 1.1 percent and an average dioxin concentration of 350 pptr. This concentration of dioxin suggests that the microorganisms' contact with the dioxin will be difficult to achieve and that the microorganisms may prefer to reduce or oxidize the more abundant and more reactive carbon compounds within the sediment, to the exclusion of dioxin, lowering the treatment efficiency.

Percentage reductions would have to be determined specifically for the matrix in question. Statements in the literature indicate that degradation in soil matrices is on the order of only a few percent over relatively large time scales, typically months. The bioreactor retention time chosen for discussion in the conceptual design of the system is a best estimate. The 30-day retention time is derived from the improvement in degradation rates noted between soil and bioslurry systems for other chlorinated hydrocarbons. It is an estimate; it is not rigorous and represents a maximum rate for bioslurry remediation to have a reasonable chance to compete with other technologies. The rate has not been measured in a bioslurry system to the knowledge of the authors.

Pretreatment requirements

The likely size of the bioslurry reactors coupled with the lengthy residence times estimated for bioremedial actions make shipboard use unfeasible. A land-based facility is the best option given these constraints. Contaminated sediment will need to be transported from the dredging location to the treatment facility. Barge transport is the reasonable alternative since contaminated sediments are assumed to exist in specific locations, requiring "batch" removal and remediation. Direct pipe transfer from nearby sites may be an alternative, depending on dredging site and treatment site locations.

Successful bioslurry operations depend on intimate contact between the contaminated particles, their surrounding fluid, and its microbial content. All these are enhanced in a relatively high solids slurry in the range of 35 percent or higher. Slurry technology works best if small particles of less than 63 µm make up the bulk of the solids. Fines are easier to keep in suspension and reduce the energy inputs that must be made to the system to maintain slurry homogeneity. Pretreatments are selected to modify the original sediment by mechanical removal of debris. Subsequently, the remaining material is partitioned by size to remove the coarse fraction which will not easily be suspended in the slurry. The coarse fraction has a low surface to volume ratio and carries a minimal amount of pollutant adsorbed to its surface. Washing of the coarse fraction removes the organic rich coating and allows it to be combined with the fines as the slurry is prepared for introduction to the bioreactor.

Posttreatment requirements

Effluent. If a 35-percent solids bioslurry is used, up to 65 percent of the volume of the slurry will be treatment water, some fraction of which will require management. It is reasonable to assume an almost total recycle of that process water back to the slurry preparation tank. The recycle loop would greatly reduce the need for large volumes of makeup water, and it would virtually eliminate the necessity of an effluent treatment unit process. The residual water content of the dewatered slurry will depend on the disposal process and its associated transportation method. There is a potential for the dewatered slurry to find terrestrial application use as fill, which may render other disposal options as unnecessary.

Runoff. Treatment and storage facility site design will have to take into account the control of runoff, especially from areas where incoming contaminated materials are handled. With the need for substantial amounts of water for slurrification, runoff may be best handled by collection in a sump system, and subsequent introduction to the slurry preparation tank. Leachate should not be an issue, given proper facility design and construction.

Air emissions. Aerobic bioslurry reactors need to be provided with a system for controlling emissions. In the case of an anaerobic bioslurry system, purging the system with nitrogen will add additional cost and complexity to the system design and operations. The bioreactor will need to be maintained at a slight positive pressure to prevent the invasion of oxygenated air during anaerobic cycles. An instrumented system will provide monitoring information on the amount of makeup nitrogen required. The offgases will be measured, and if necessary collected and passed through an appropriate scrubber to provide control of potential emissions.

Chemical analysis for dioxins will add another significant cost to the process. Costs for trained labor for sample capture, shipment and data handling, and actual analytical costs are not inexpensive for these contaminants. Onsite analytical capability may prove to be beneficial to provide the information necessary for process control. Independent confirmation analysis may be required before disposal.

Feasibility of full-scale implementation

Processing equipment for this treatment can be constructed from off-the-shelf items. Bioslurry reactors are available from various suppliers, and the materials handling equipment is conventional and available. Setup time for a commercial operation is unknown, but can be expected to be less than 1 year, considering the ease of availability of equipment.

Pretreatment transportation needs and storage volumes are dependent upon the amount of contaminated sediment that is to be processed. The dredging and transport of the sediment to the storage and treatment site can be accomplished at rates far in excess of the processing rate of this treatment. Therefore, the storage volume required is a function of the volume of sediment to be dredged. The land area required for the process equipment is difficult to determine accurately at this stage of development, but it is not expected to exceed 5 acres, in addition to the anaerobic ponding storage area. This would provide all the space necessary for bioreactors, support equipment, dewatering equipment, and material handling equipment.

Effluent volumes are expected to be small, a result of the continual recycling of process water into the slurrying process. Air emissions are not expected to be a major concern. Conceptually, the process will reduce the contaminants at least to simple nontoxic hydrocarbons, or to carbon dioxide and water, leaving no hazardous residue for disposal or further treatment.

Limiting maximum concentrations of contaminants have yet to be defined. For this bioremediation process, minimum contaminant concentrations are more important, especially when applied to typical sediment concentrations. The technology is insufficiently developed to address either limit.

Because the influent stream is required to be greater than 65 percent water, this process is not sensitive to sediment characteristics except that particle size is controlled by screening as a method of pretreatment volume reduction.

Human health and environmental impacts related to the process do not appear to be significantly greater than for normal industrial processes involving the movement of large volumes of soils or sediment. No toxic chemicals are required for the treatment. Nor are high temperatures or high pressures required. The contaminant level of the untreated waste would be a source of greater risk than any of the other products from the process. Effluent and final product analyses will be required to ensure the process is operating correctly.

Regulatory requirements will be site specific, depending on zoning area, local government, and state government controls. Federal regulations can also vary with the site location and level of contamination. If the sediment is controlled under TSCA or RCRA regulations, specific procedures for analysis of the effluent stream may be mandated to prove that contaminant concentrations are below levels of concern and can be "delisted" for conventional disposal.

Public concerns are difficult to forecast at this time. This process does not require procedures that have proved unpopular with the local communities at other treatment sites, such as incineration emissions, residual hazardous material disposal, or hazardous chemical spills.

Processing rates cannot be determined at this stage of development of the technology. However, it can be assumed that remediation will proceed significantly more slowly than with any other technology. Several months or years may be required to attain complete contaminant removal within each bioslurry reactor. A recent estimate for biological degradation of PCB-contaminated sediments suggests 2 to 3 years would be necessary for a single-cycle anaerobic-aerobic system to reduce contaminants by 50 to 60 percent.[1] Multiple bioreactor units could decrease the overall treatment time.

Costs

Estimated costs for this treatment technology will be made up of a large number of component costs for site, processing equipment, material handling equipment, and personnel support and monitoring equipment. Given the uncertainties regarding details of the actual treatment train, an overall planning estimate of the costs associated with the treatment process has been based on Yang et al. (1987) and the Site Applications Analysis Report for the BioTrol Soil Washing System (USEPA 1992d).

Yang et al. (1987) has developed capital costs for a rotating biological treatment system for wastewaters. This system is similar to the proposed bioslurry unit in that it requires a number of unit operations such as pH neutralization, nutrient addition, clarification, sludge dewatering, which are comparable to wastewater treatment. BioTrol, Inc. (Princeton, NJ), developed an economic analysis based on a SITE demonstration at New Brighton, MN. The process consisted of soil washing, biodegradation of the organic contamination on the soil fines in a slurry reactor, and a biological treatment of the process water recycled from the soil washing operation (USEPA 1992d). Estimated costs for a full-scale commercial application were derived from the appropriate costs per cubic yard and costs per month and were extrapolated to a conceptual dioxin-contaminated sediment treatment.[1] Best estimates for storage and anaerobic bioremediation treatment within a CDF for 12 months, as described by Hughes, were made. Bioslurry treatment (either anaerobic or aerobic) costs were estimated over an additional 12, 18, or 24 months, depending

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[1] Memorandum, 1993, D. Hughes, Hughes Consulting Services, Syracuse, NY.

on the volume of sediment treated. A mobile treatment system is assumed to be the most cost effective. Fixed costs were estimated from BioTrol's figures, adjusted for sediment volume. Variable costs were derived from both sources, adjusted for either time of processing or volume to be processed. For the first 12-month period, when the sediment remains under anaerobic conditions in a CDF, labor and equipment costs were assumed to be low, relative to the bioslurry treatment phase. For this conceptual process, it is assumed that the treated sediment will be disposed of at a landfill separate from the treatment site. Table 18 summarizes the cost analysis.

Stabilization/Solidification Treatments

Description of treatment process

The process of treating waste by stabilization/solidification (S/S) methods has been used for over 20 years to treat a variety of wastes, including chemical- and radioactive-contaminated materials. The S/S process works by restricting the mobility of the contaminants. This is accomplished by binding the contaminants into a monolithic mass that has low leachability, low reactivity, and low free liquid content. In addition, the solidified mass must possess physical stability and strength and be highly resistant to erosion and biodegradation. There are several binding agents that can be used to create a solidified mass from contaminated materials. These include, Portland cement, lime plus pozzolans (fly ash, kiln dust), thermoplastics (bitumens), and thermosetting organic polymers (epoxies). When added to the contaminated material in the appropriate proportions, these additives will form the required solidified mass. Combinations of additives also have been shown to be effective. Another process that has been tested in recent years is vitrification. In this process, the contaminated waste is heated until it melts and then cooled into a glass-like mass.

Treatment with lime plus pozzolans. This S/S treatment combines lime (CaO) and pozzolans with the contaminated material to form a monolithic mass. A pozzolan, as defined by American Society of Testing and Materials Standard Number C618, is as follows:

> "a siliceous or siliceous and aluminous material, which in itself possesses little or no cementitious value, but will, in finely divided form and in the presence of moisture, chemically react with calcium hydroxide at ordinary temperatures to form compounds possessing cementitious properties. There are several naturally occurring materials that can be used as pozzolans. These include, but are not limited to, shales, opaline cherts, tuffs, pumicites, and diatomaceous earth. In addition, some artificial materials such as fly ash and kiln dust exhibit pozzolan type properties."

Table 18
Cost Analysis: Bioremediation Process

Item	Cost, \$/yd^3	Type of Cost	Present Value Cost \$,000s for		
			10,000 yd^3	50,000 yd^3	100,000 yd^3
Site preparation		Fixed	375	375	375
Permitting and regulatory costs		Fixed	1	1	1
Capital equipment	20	Variable	200	927	1,685
Start-up and fixed costs		Fixed	250	250	250
Pretreatment of wastes		Variable[1]	0	0	0
Labor costs	80	Variable	783	3,619	6,582
Consumables, supplies, and utilities	24	Variable[1]	235	1,086	1,974
Effluent treatment and disposal		Variable[1]	0	0	0
Monitoring and analytical costs	13	Variable	127	588	1,070
Maintenance and repair costs	16	Variable	156	724	1,316
Site demobilization and cleanup	7.5	Fixed	96	82	67
Dredging	15	Variable	75	375	750
Transportation to TSF		Variable	150	750	1,500
Construction of TSF[2]		Variable	300	1,000	1,800
Land lease for TSF[3]		Variable	12	115	524
Land lease for process equipment[3]		Variable	6	29	52
Transport of residual wastes	67.28	Variable	90	405	714
Disposal in a landfill		Variable	658	3,043	5,535
Total, \$,000s			3,426	12,962	23,483
Cost			\$370	\$285	\$270
Throughput (feed rate) in yd^3/day			30	30	30
Period for completion (in months) (using 5 working days per week)			10.9	54.6	109.3
Monthly discount rate (assuming 4.4% annual discount rate)	0.37%				

[1] These costs are combined.
[2] These costs are estimated at \$30/yd^3 for 10,000 yd^3, \$10/yd^3 for 50,000 yd^3, and \$18/yd^3 for 100,000 yd^3.
[3] Land lease costs are based on an annual rate of \$7,000/acre.

These materials are mixed into a slurry type material, in situ or in a concrete batch plant. This slurry is then either left in place to cure or is transported to a remote curing area. During the curing process, the slurry hardens into a monolithic mass. This mass is not of sufficient strength to withstand the weathering action it would be subject to in an exposed situation. Therefore, it would have to be transported to a landfill for disposal (Figure 17).

Some experience with PCB-fixation has been gained by the use of proprietary additives. International Waste Technologies (Wichita, KS) has performed laboratory tests and a SITE demonstration where the cementing additives were mixed in situ with contaminated soils at the demonstration site. Immobilization of PCBs appeared to be successful, but could not be confirmed because of low concentrations in the untreated soil. Leachate testing performed one year later on treated soil samples showed no increase in PCB concentrations, suggesting immobilization (USEPA 1991b). Results obtained from bench-scale tests of the treated material indicated that the maximum detectable leachable dioxin concentration after treatment with this process was 10.4 pptr, and 2,3,7,8-TCDD concentrations were measured at 3.0 pptr (U.S. Congress 1991).

Treatment with Portland cement. This S/S method utilizes Portland cement to form an inert, impervious, durable mass. Portland cement concrete is comprised of three basic components: Portland cement, aggregate, and water. Of these three components, two (water and aggregate) are contained in the New York/New Jersey Harbor materials. All that remains is to adjust the amounts of these materials to the correct mix ratio, and add the proper amount of Portland cement. The most efficient water to cement to aggregate ratio would need to be determined by treatability tests. Once this ratio is determined, these components would be mixed in an industrial concrete mixer and handled as any normal concrete. This material could then be poured into concrete forms or placed in a disposal facility and subjected to the normal concrete curing process.

Depending upon the sediment characteristics, dredging method and water content at treatment, this process may not require dewatering.

Treatment with thermoplastics. This S/S treatment utilizes thermoplastics, most notably asphalt, to immobilize contaminated material. Unlike the lime and pozzolan mixture, asphalt needs a relatively low moisture content (less than 5 percent). A rotary kiln would be required to achieve this level of drying. The dried aggregate would then be fed into a continuous mix asphalt plant. This asphalt mixture is then cooled and subjected to leaching exposure testing to determine the appropriate disposal method. In addition to the final asphalt cement product, the offgas from the rotary kiln would require monitoring and probably treatment to remove volatile organics (Figure 18).

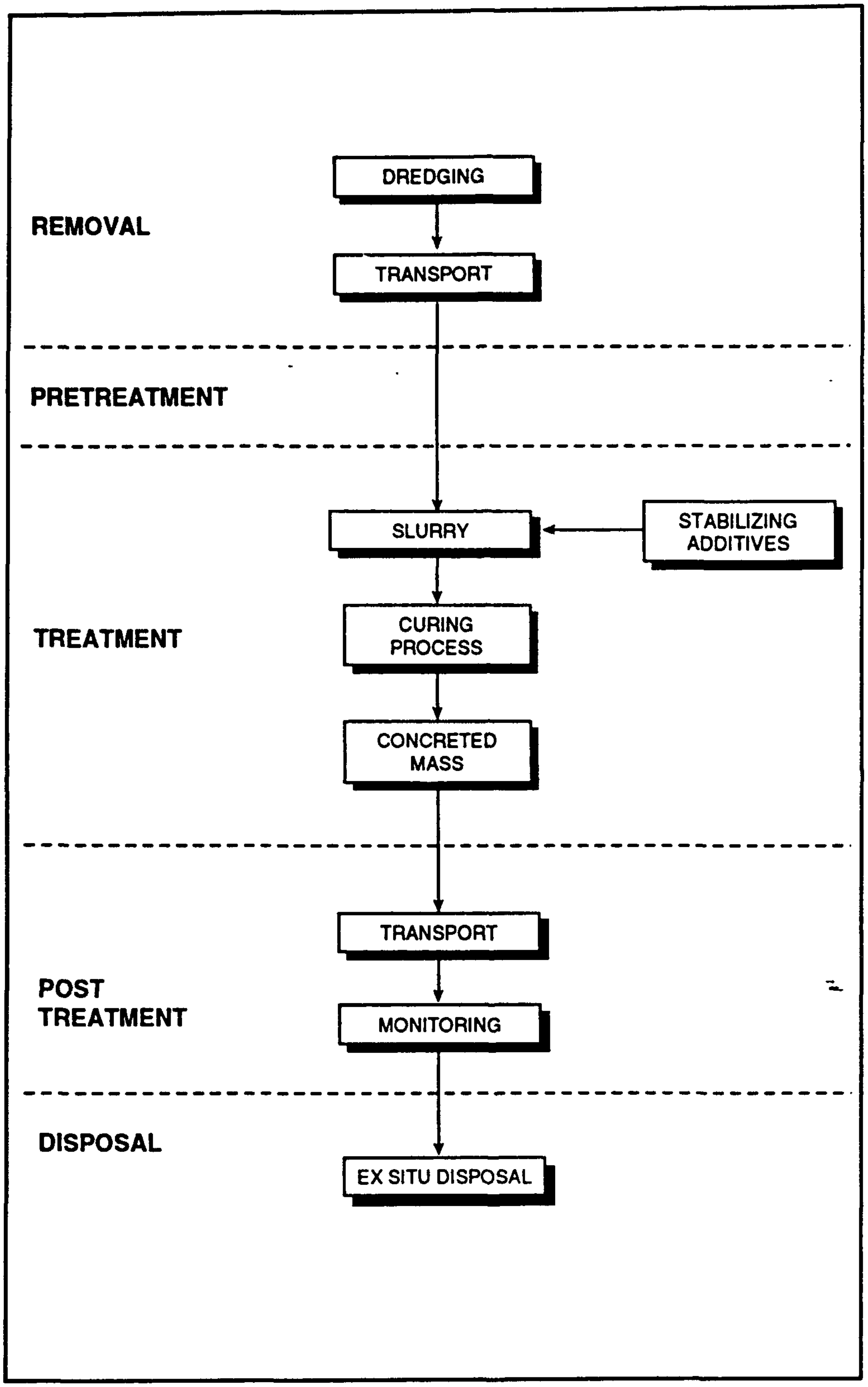


Figure 17. Process flowchart for lime or Portland cement stabilization/solidification treatment

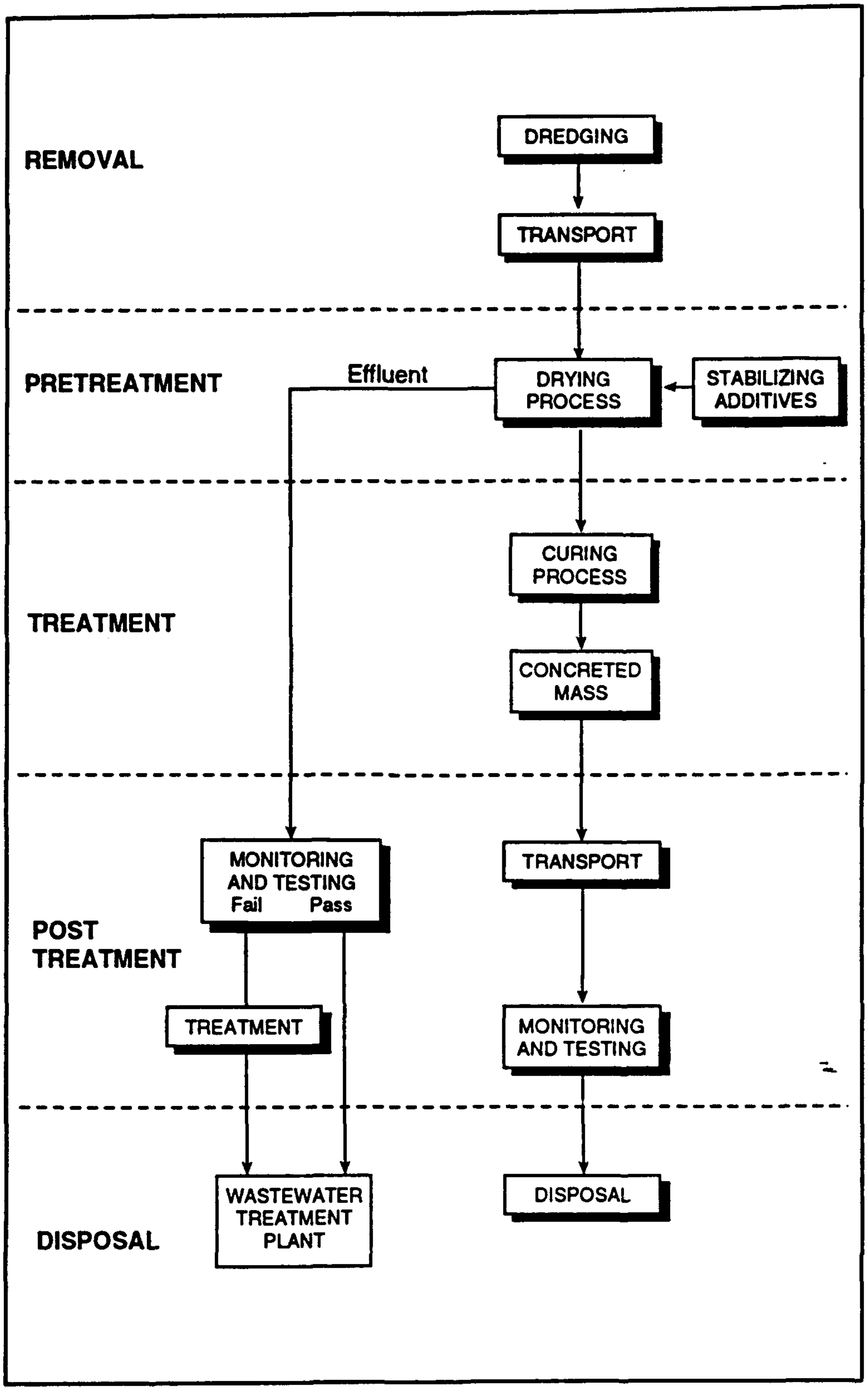


Figure 18. Process flowchart for thermoplastic stabilization/solidification treatment

Treatment by vitrification. Vitrification can be used to treat soils and sludges contaminated with mixtures of various waste types (e.g., radioactive, inorganic, and organic compounds). The process converts contaminated material into a durable glass and crystalline form. During the process, heavy metals and other inorganic constituents are retained and immobilized in the glass structure; organic constituents are typically destroyed or volatilized and captured by an offgas treatment system. Example proprietary processes are in situ vitrification offered by Geosafe Corporation (Kirkland, WA) and plasma arc vitrification offered by Retech, Inc.

The in situ vitrification process could be applied to sediment that has been dredged from the waterway and placed into a confined disposal facility. It would not be appropriate for vitrifying bottom sediments in the waterway. Four electrodes connected to a utility distribution system or to an onsite diesel generator are inserted into the soil. As the current flows between electrodes, the adjacent soil is heated to 1,600 to 2,000 °C, well above the typical melting temperature of soil. Within the melt, organic contaminants are vaporized and pyrolyzed (i.e., thermally decomposed); the pyrolysis products rise to the surface and combust in the presence of oxygen. Nonvolatile inorganic constituents are dissolved or incorporated into the melt. Volatile metals may vaporize and rise to the surface along with the pyrolysis products. After cooling, a vitrified monolith is created, with a silicate glass and microcrystalline structure.

A negatively pressurized hood placed over the process area collects both organic and inorganic gases, which are treated before being released into the atmosphere. An offgas treatment system can be designed to handle conditions at most sites and may include any of the following units: a wet scrubber system, a heat exchanger with a glycol cooling system, a heater, a filter, and/or an activated charcoal assembly. The hood draws in large amounts of outside air, which helps to oxidize combustible vapors and pyrolysis products. All equipment involved with the vitrification process, including the offgas treatment system, are contained in mobile trailers.

Following treatment, the surface of the vitrified area is covered with clean soil, and the melt is allowed to cool slowly, producing an amorphous solid resembling obsidian (Figure 19). Several months are required for the treated area to cool to ambient temperature; however, after 4 to 5 days, the melt has cooled sufficiently for equipment to be moved onto the treated area (USEPA 1991b).

State of development

For years, lime with fly ash S/S systems have been successfully used in managing hazardous wastes. However, the containment performance generally is such that a hazardous waste would still be classified as hazardous after processing. That is, the destruction or immobilization efficiency of

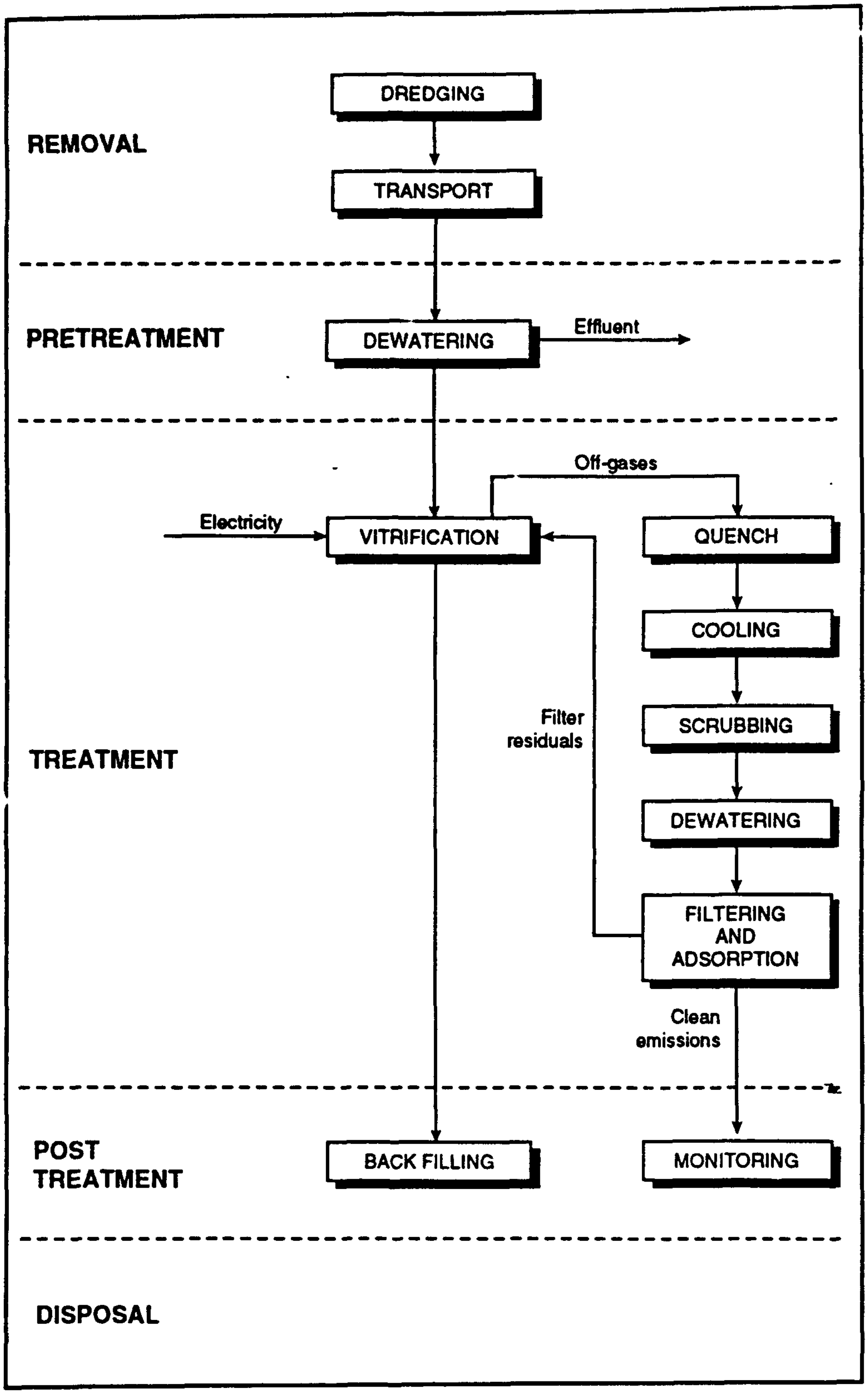


Figure 19. In situ vitrification process flowchart

the processes is not high. Lime/fly ash-sorbent-based landfills have been established using liner and monitoring systems to ensure safe disposal (USEPA 1986b). When considered as a total treatment system, S/S with lime and landfilling can successfully confine contaminants by eliminating migration, via water and wind, into the surrounding environment.

A number of commercial solidification vendors are currently operating using variations of Portland cement systems. Many use specific sorbents, additives, and proprietary formulations developed to deal with the needs of specific wastes. Processes of this type have been tested at sites around the country. This method has been used as a temporary solution to contaminant immobilization in past applications, but the technology has not demonstrated the equivalent of the 99.9999-percent removal or immobilization standard.

A combination of methods were tested at the laboratory scale on dioxin-contaminated soils from three sites in Missouri. Concentrations of 2,3,7,8-TCDD ranged from 32 to 700 ppb in untreated soils. Cement and soil mixtures were found to be susceptible to breakdown by weathering. A combination of emulsified asphalt and lime added to the soils was found to be most successful. Leachability testing of this mixture indicated that dioxin immobilization was successful, at least to the detection limit of the test procedure (U.S. Congress 1991).

At least seven S/S techniques are being evaluated within the USEPA SITE demonstration program. Some of these may prove effective when used in conjunction with other treatment technologies for contaminated sediments by stabilizing residues from these processes (U.S. Congress 1991).

Vitrification technologies have been tested on several different soil types containing heavy metals, liquid and solid organics, and radioactive materials. Over 130 bench-, field-, and large-scale tests have been performed. Bench-scale treatability tests have been carried out on PCB-contaminated sediments from the New Bedford Harbor Superfund site and on PCB-contaminated soils at other Superfund sites (Liikala 1991). This technology has been selected as a preferred technology at 10 private, USEPA Superfund, and Department of Defense hazardous waste sites (Hansen and Fitzpatrick 1991).

Mixtures of contaminants within the soil or sediment do not affect the efficiency of the treatment. Contaminant concentrations in the untreated material also are not limiting to the process. Initial PCB concentrations of greater than 19,000,000 ppb (19 g/kg) have achieved typical destruction efficiencies of 99.9 to 99.99 percent. When combined with the removal fraction (from the offgas after destruction in the pyrolysis process) of 99.9 percent, the overall DRE reaches the USEPA incineration standard of 99.9999 percent. Similar DREs from bench-scale tests with untreated dioxin concentrations of greater than 47,000 ppb and furan concentrations higher than 9,400 ppb have been reported also (Hansen and Fitzpatrick 1991).

Tests have also indicated that migration of PCBs into the surrounding soil did not occur as a result of the vitrification process. The possibility of the creation of dioxins and furans in the adjacent soil from the dechlorination of PCBs (as opposed to the destruction of PCBs) has been investigated also. Engineering-scale tests (0.05 to 1.0 tons) indicate that PCBs are destroyed and no dioxin/furan compounds are created. The tests have shown also that low dioxin/furan concentrations present in the surrounding soil (soil that is heated but not vitrified) before the treatment were reduced when measured after the testing (Liikala 1991).

Significant volume reduction (25 to 40 percent for most soils) occurs as the solid particles melt and the interstitial voids are removed.

Pretreatment requirements

The preferred method of sediment removal is with a clamshell dredge. This process yields the lowest water content, which will minimize the amount of cementing agents required or the energy input for vitrification. There is no preferred method of transportation to the treatment facility.

In general, minimal pretreatment is required for stabilization processes based on the addition of cementing or fixing agents. As is necessary for all treatment technologies, the material will need to be tested for compatibility with the treatment process. A number of materials (such as sodium borate, calcium sulfate, potassium bichromate, and carbohydrates) can interfere with the chemical reactions. Oils and greases can also physically interfere with bonding by coating waste particles. The cementing system is strongly alkaline and can react with certain waste to release undesired materials such as gas or leachate (USEPA 1986b).

Dewatering is potentially a beneficial pretreatment process, but a detailed cost analysis will need to be performed to determine the appropriate reduction of water content before treatment.

For Portland cement additives, compatibility testing will investigate effects of the waste characteristics on the setting times and stability of the silicates and aluminates that form when the cement hydrates. Additionally, other materials such as oil and grease or large amounts of soft, fine wastes can prevent bonding of particles in the waste and can lower the ultimate strength of the treated material. Acidic or acid-producing materials such as sulfides can react with carbonate and hydroxides and destroy concrete after setting has occurred.

In addition, the very high alkalinity of hydrating Portland cement can cause the evolution of ammonia gas if ammonium ions are present in abundance in the waste. Some metals have increased solubility at the very high pHs that occur in the cement hydration reaction (e.g., nickel, lead, and zinc) (Figure 20).

Pretreatment requirements for asphalt-containing additives are more complex. The contaminated material will need to be dewatered to a low moisture content before it can be combined with the asphalt binder. Up to 15 percent calcitic lime is required to improve the stabilization characteristics of the treated matrix.

Compatibility of the waste and the solidification matrix becomes a major consideration when using thermoplastic additives. Most matrices used with contaminated wastes are reduced materials (solid hydrocarbons or sulfur) that can react (combust) when mixed with an oxidizer at elevated temperatures. Other compatibility problems are related to unusual softening or hardening of the waste/matrix mix. Some solvents and greases can cause asphalt materials to soften so that they never become rigid solids. Borate salts can cause hardening at high temperatures and can stall or clog mixing equipment.

Salts that are dehydrated at the elevated temperatures used in mixing can interfere with the stabilization process. Sodium sulfate hydrate, for example, will lose water during asphalt incorporation; and if the waste/asphalt mix containing the dehydrated salt is soaked in water, the mass will swell and crack because of rehydration.

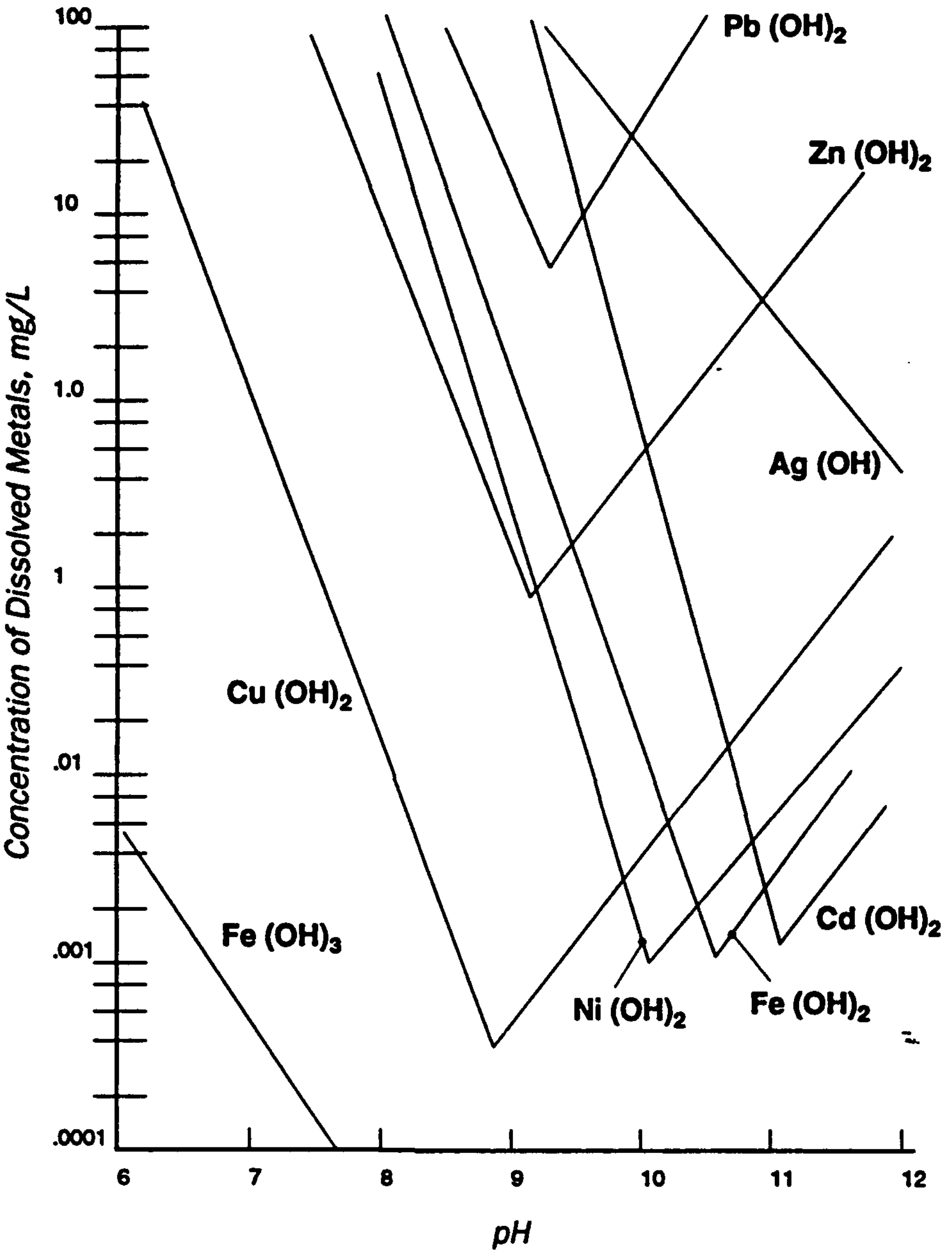
Vitrification requires that the contaminated waste must contain a sufficient amount of glass-forming material, in the form of nonvolatile, nondestructible solids to produce a molten mass. The process is limited by large void volumes (greater than 150 ft^3), debris exceeding 20 percent, and combustible organics exceeding 5 to 10 percent by weight (USEPA 1991b). Sediments will not usually exceed these limits; however, if they should, remedial pretreatments would be required.

Since a significant amount of energy is required to vaporize water (approximately the same amount as is required to melt soil on a per weight basis), it is economically beneficial to minimize the water content of the sediment. Processing rates can be affected also by the moisture content.

Posttreatment requirements

Stabilization processes using cement additives do not generate effluent, runoff, leachate, or air emissions.

Portland cement-based S/S systems have proved to be some of the most versatile and adaptable processes for contaminant immobilization. Waste and concrete composites can be created that have exceptional strength and excellent durability, and that retain wastes very effectively. The addition of selected sorbents and/or emulsifiers often overcomes the problem of pollutant migration through the somewhat porous solid matrix and



Source: USEPA. 1987. Office of Solid Waste and Emergency Response, EPA/625/6-87/6-87-015.

Figure 20. Experimentally determined solubilities of metal hydroxides

consequently lowers the leaching losses from the treated wastes (USEPA 1986b). Materials stabilized with Portland cement are more likely to pass the Toxicity Characteristics Leaching Procedure (TCLP) testing and may not require disposal in a controlled landfill.

Thermoplastic processes require low-moisture content feedstock material. The drying process will be a possible source of effluent and/or air emissions. If a drying kiln is used, emissions will require monitoring and possible further treatment before release. If mechanical dewatering techniques are used, effluent monitoring and control will be necessary. Once treated and stabilized, this material will be relatively stable except for the possible swelling and cracking because of the salt content of the dried aggregate. If this occurs, the material may fail TCLP tests and require disposal in a controlled landfill facility.

Feasibility of full-scale implementation

Stabilization/solidification processes that use cement or lime as additives are not proprietary; however, some additives to the concrete mix may be proprietary. A general conceptual installation will require the following:

- A concrete batch plant and operating crew.
- A front-end loader and operator.
- Two to six laborers.
- Concrete forms and curing equipment.

The treatment process would require the use of a concrete batch plant and approximately 1 acre of land on which to erect and operate the plant. The curing, loading, and unloading processes will require an additional 1 to 3 acres of land. In all, approximately 2 to 4 acres of land in an industrial area will be needed. The curing process will require approximately 8 day's worth of treated materials to be stored onsite at any one time. Once the material has been treated, it can be loaded onto trucks and transported to the final disposal site or contained permanently at the treatment site.

Safety is of major concern at any construction site. As dioxin is primarily an inhalation/ingestion hazard, it should not prove to be an additional hazard as long as the untreated material is not dried to such an extent that airborne particles are created.

The immobilization process using an emulsified asphalt mixture necessitates the use of a dewatering device, a rotary kiln, and an asphalt batch plant. This entire process will demand approximately 2 acres of land to erect and operate. The curing, loading, and unloading process will require the use of an additional 1 to 2 acres of land. In all, approximately 3 to

4 acres of land will be required. This land should be located in an industrial area. The curing process will require 1 day's worth of treated material to be onsite at any one time. Once the material has been treated, it can be loaded onto dump trucks and transported to the final disposal site, or the CDF could be converted to a beneficial use once it reaches capacity.

This process is currently available and is not a proprietary process; however, some additives to the asphalt mix may be proprietary. This process is made more hazardous than the concrete stabilizing processes because at one point in the process, the contaminated material requires complete dewatering. Effluent and emission monitoring control equipment will be needed to monitor and treat the substantial waste stream created by the pretreatment dewatering.

The in situ vitrification process equipment can be mounted on three trailers and can be quickly mobilized. Once at the treatment site, operations can commence within a few days. This process is commercially available from Geosafe Corporation, through a license to the U.S. Department of Energy. At this time, a single full-scale commercial system is available. This system can process between 13,000 to 22,000 yd^3 of soil per year.

Because this is an in situ process where the treatment equipment moves over the treated material, the area of land required is controlled by the volume of material to be treated. The most efficient material depth for treatment is 20 to 30 ft. At a stockpile depth of 20 ft, 100,000 yd^3 of sediment would occupy approximately 3 acres. Thus, a 5-acre CDF would be of adequate size to store and treat up to 100,000 yd^3 of material at one time.

Transport of the untreated material to the treatment site will be necessary. Because the treatment creates in situ homogenized monoliths, it would not be practical to move the treated material. This implies that, on a large scale, each treatment site also becomes the final storage site for the material. If further transportation was necessary, the monoliths would have to be broken into transportable sizes. Because the treatment results in substantial volume reduction, the treatment site could be backfilled with clean material and opened for an appropriate beneficial use.

The equipment required for the treatment process includes the three trailers of control and processing and emission control equipment, an offgas collection hood, which is transported on one of the trailers and assembled at the site, a crane and a tractor to move the offgas hood and trailers between melts, and a diesel-powered generator or access to an electricity supply. The offgas treatment system uses conventional unit process equipment that is required by any thermal-treatment process that must treat offgases. This system is technologically simple so that adequate margins of safety and reliability are easily obtained. The onsite and in situ nature of the process and the quality of the residual product have resulted in generally excellent acceptance by the public (Hansen and Fitzpatrick 1991).

Costs

This cost analysis (Table 19) is based on data collected from a USEPA SITE demonstration project of the Solidtech, Inc., Stabilization/Solidification Process (USEPA 1990c). This process blends waste material with pozzolanic material (such as fly ash), kiln dust, or cement; water; proprietary additives; and Urrichem, a proprietary reagent. The process equipment, including a mixer, is readily transportable on one or two trailers. The equipment is self-contained and requires minimal setup time. Two people are required to operate the equipment. Other personnel are required for support activities such as quality control, chemical formulation, and office support. Personnel are also required to load the waste material and remove the treated waste.

It is assumed for this cost analysis that the treated sediment will remain permanently at the CDF/treatment site. Once the treatment is complete, or more likely, once the CDF has reached capacity, the site will be capped with clear soil and vegetated. The land is assumed to be purchased at a cost of $70,000/acre. Two acres are assumed for $10,000 per cubic yard, four acres for $50,000 per cubic yard, and 10 acres for $100,000 per cubic yard. No long-term monitoring of runoff or leachate is considered, nor are the costs of converting the site to a beneficial use.

Table 19
Cost Analysis: Stabilization/Solidification Process

Item	Cost, $/yd^3$	Type of Cost	Present Value Cost $,000s for		
			10,000 yd^3	50,000 yd^3	100,000 yd^3
Site preparation		Fixed	30	30	30
Permitting and regulatory costs		Fixed	12	12	12
Capital equipment	16	Variable	161	772	1,466
Start-up and fixed costs		Fixed	32	32	32
Pretreatment of wastes		Variable[1]	0	0	0
Labor costs	60	Variable	590	2,825	5,373
Consumables, supplies, and utilities	77	Variable	758	3,636	6,907
Effluent treatment and disposal	1.6	Variable	15	73	139
Monitoring and analytical costs	7.7	Variable	76	366	696
Maintenance and repair costs	0.4	Variable	4	20	38
Site demobilization and cleanup		Fixed	17	16	14
Dredging	7.5	Variable	75	375	750
Transportation to CDF	15	Variable	150	750	1,500
Construction of CDF[2]		Variable	300	1,000	1,800
Land purchase for CDF[3]		Variable	138	265	630
Land purchase for process equipment[3]		Variable	70	70	70
Transport of residual wastes		Variable	0	0	0
Disposal in a landfill	0	Variable	0	0	0
Total, $,000s			2,429	10,246	19,458
Cost			$242	$205	$195
Throughput (feed rate) in yd^3/day			80	80	80
Period for completion (in months) (using 5 working days per week)			5.8	28.8	57.7
Monthly discount rate (assuming 4.4% annual discount rate)	$0.37%				

[1] These costs are implicit in other costs.
[2] These costs are estimated at $30/$yd^3$ for 10,000 yd^3, $20/$yd^3$ for 50,000 yd^3, and $18/$yd^3$ for 100,000 yd^3.
[3] Land purchase costs are estimated to be $70,000/acre.

4 Disposal and Containment Options

General Description of Dredged Material Disposal Operations

Two dredged material disposal options have been mainly utilized in the New York area: unrestricted open-water disposal and disposal with capping. Between 80 and 90 percent of the dredged material from the Port of New York and New Jersey is disposed at aquatic sites (New York University 1989). Material classified as uncontaminated by present regulatory guidelines is acceptable for unrestricted aquatic disposal. This operation consists of simply dredging the material and placing it at the USEPA-designated Mud Dump Site, or some other approved site. No special conditions, handling, management, treatment, or other restrictions are imposed on the operation. Dredged material with limited contamination can be disposed at the Mud Dump Site if it is subsequently covered, or capped, with a layer of uncontaminated material. This capping layer confines the contaminants and inhibits their dispersion into the surrounding environment. A third disposal option that has not been used to date, but has been considered, is disposal of contaminated sediment in a containment island, which is a diked confined disposal facility constructed offshore.

Unconfined Open-Water Disposal

Description of the disposal option

Unconfined open-water disposal has been the routine disposal method for the majority of material dredged from the New York/New Jersey Harbor area. The process is straightforward and does not involve any treatment of the dredged material. Dredging is carried out by hopper dredges or hydraulic dredges filling barges, or more often with a clamshell bucket operated from a barge-mounted crane. Sediment is excavated with the bucket and placed in a scow or barge for transportation by tug to the disposal site. When the tug and barge reach the disposal site, the bottom of the barge is opened, and the dredged material falls to the sea floor. Typically, several barges are used so that the dredge can operate continuously: as one barge is towed to the disposal site, another is being filled. The procedural flowchart for unconfined open-water disposal is shown in Figure 21.

State of development

Unconfined aquatic disposal is the standard and most widely used method of disposal of dredged material not only in the New York area, but throughout the world. The process is applicable to any volume of sediment with wide range of physical characteristics. For uncontaminated sediments, the process is considered environmentally sound.

The method of removal of the sediment from the bottom (dredging) has limited impact on the disposal of the dredged material. Prior to disposal, representative samples of the sediment must be collected for physical, chemical, and perhaps biological analyses to characterize the material and to determine if it can be considered uncontaminated.

Transport to the disposal site is generally by split-hull barges. No special navigation capabilities are required because the location of the Mud Dump Site is identified by a permanently moored marker buoy. Aquatic disposal creates some transient increase in suspended solids in the surrounding water column immediately after discharge of the dredged material. Monitoring at the disposal site is required on a regular basis to check adverse environmental impacts of the disposal operations and the accumulation of dredged material on the sea floor.

Feasibility of full-scale implementation

The feasibility of performing unconfined aquatic disposal of dredged material is demonstrated by its routine use throughout the world. Transport barges are needed to store and carry the dredged material to the disposal site. The number of barges required is a function of the dredging capacity and the travel time from the dredging site to the disposal site. No construction at the dredging site is typically required, and no postdisposal cleanup is usually necessary.

Regulatory classifications of contaminated sediments are based on the probable environmental effects of the sediment when disposed. If biological test results of sample sediments exceed predefined limits, such as elevated containment concentrations in tissues of aquatic organisms, then the dredged material cannot be classified as uncontaminated and thus cannot be disposed in an unconfined manner.

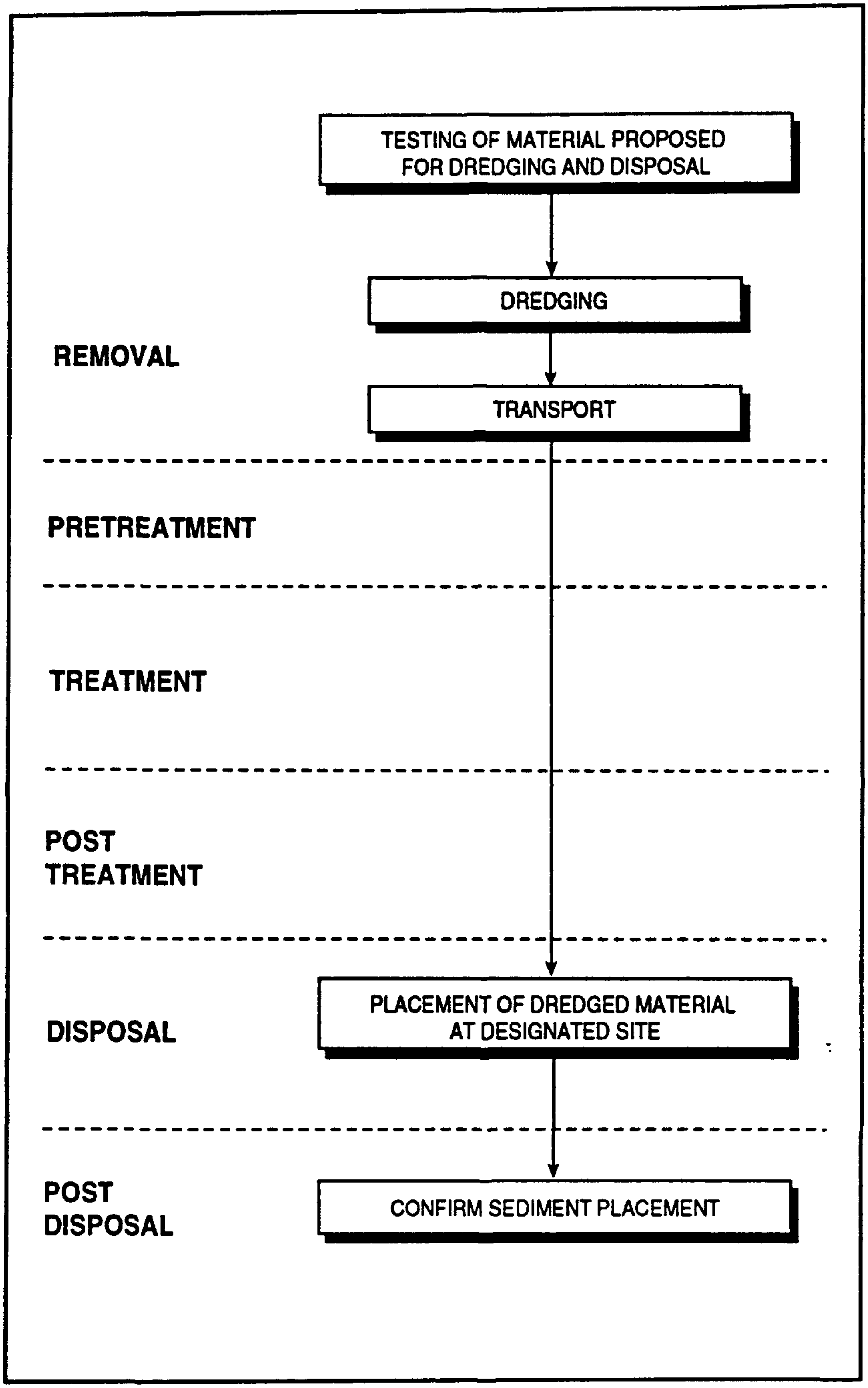


Figure 21. Unconfined aquatic disposal process

No special equipment is normally required beyond that necessary for conventional dredging, transport, and disposal. Because only uncontaminated material may be approved for unconfined aquatic disposal, special safety precautions are not required. Many commercial dredging and maritime transport companies with extensive experience and equipment are established in all major ports, making the process readily available.

The disposal site designation process usually results in the development of an environmental impact statement (EIS). Integral procedures in the approval of the EIS include community meetings at which details of the EIS investigation are discussed and periods of time in which public comments on the draft EIS are solicited.

Permitting

The Marine Protection, Research and Sanctuaries Act requires that a permit be issued by the Corps of Engineers for all dredged material disposal projects at a designated ocean disposal site located in the territorial sea and all ocean waters beyond. This Act also controls the types of materials that may be disposed at sea. The Clean Water Act addresses some instances of permitting for dredged material placement within the territorial sea.

Costs

For open-ocean disposal, no land-based processes are required, and no costs associated with the lease or purchase of land are incurred. Beyond the standard dredging equipment and barges for transport, no special equipment or installations are required. Annual operating costs at an open-water disposal site would include the maintenance of marker buoys, periodic monitoring of the water quality, benthic communities, and sediment quality, and the costs of surveillance and enforcement of disposal permit conditions. Maintenance and repair of the transporting barges and tugs could also be included in the operating costs.

The average cost of dredging and unrestricted aquatic disposal in the New York/New Jersey Harbor area is approximately $6.00 to $7.50 per cubic yard for uncontaminated sediments. Disposal costs only are estimated to be about $3.00 to $4.00 per cubic yard.[1] Costs for individual disposal operations will vary somewhat, depending on the distance from the dredging project to the disposal site. Mobilization and demobilization costs for dredging and transportation equipment reflect the quantity of material to be dredged and disposed, the dredging depth, and the sediment characteristics; the greater the capacity of the dredging equipment, the

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[1] Personal Communication, 9 December 1992, H. Hawkins, Navigation Branch Chief, New York District.

greater the number of barges required for continuous operation. Table 20 summarizes the costs for three different volumes of dredged materials. Costs were based on an average distance between the dredging and disposal sites of 20 miles, and assumed overflow of dredged material from the barges was permitted. When overflow is allowed, the barge can be loaded with more material; therefore, fewer barge trips to the disposal area are required.

Table 20
Open-Water Dredging and Disposal Costs

	Cost		
Item	**10,000 yd^3**	**50,000 yd^3**	**100,000 yd^3**
Dredging and transportation to disposal site[1]	$75,000	$340,000	$600,000
Mobilization/demobilization	$25,000	$75,000	$100,000
Total	$100,000	$415,000	$700,000

[1] Assuming 20-mile distance and barge overflow operations; if barge overflow is not used, costs for this item would double.

Processing rates again vary depending upon physical and regulatory conditions at the dredging site. Compacted sediments and confined spaces that require frequent dredging barge repositioning will generally slow production and require smaller capacity dredges or clamshell buckets. Overall, dredging and disposal rates can range from 3,000 to 20,000 yd^3/day.

Subaqueous Capping of Disposed Material

Description of the disposal process

Capping is the burial of dredged material containing contaminants with stable layers of clean dredged material in either borrow pits or on the open ocean floor (O'Connor and O'Connor 1983). The objective of the cap is to isolate the water column, as well as biota, from contaminants present in the dredged material. Capping has become a viable alternative for the management of dredged materials that do not meet the regulatory criteria for unrestricted subaquatic disposal. It has been used extensively in recent years, with particular application in the New York and New England area. A flowchart illustrating the capping process, from sediment removal through disposal, is presented in Figure 22. Considerations for capping site selection are discussed below.

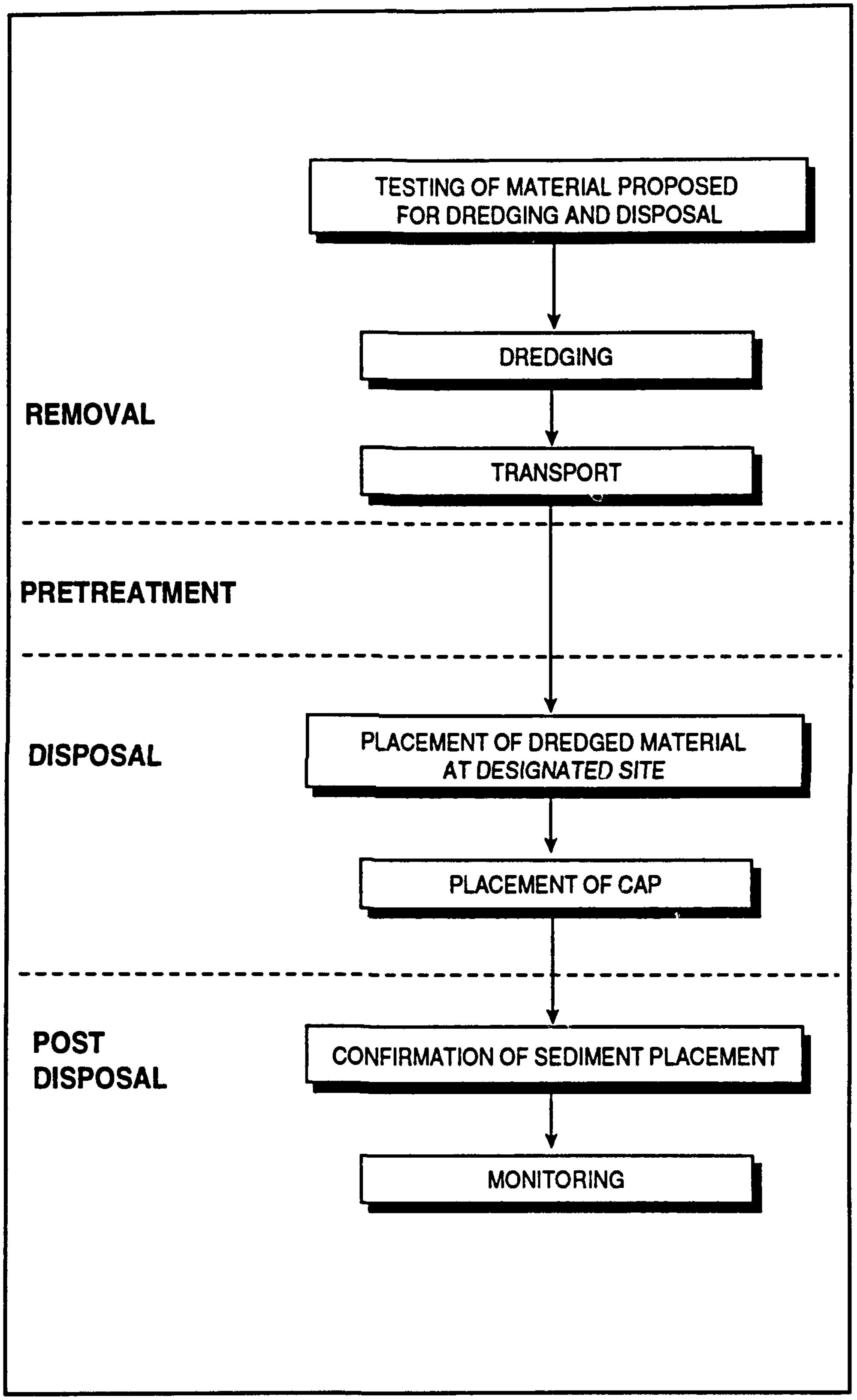

Figure 22. Subaqueous capping process

Physical site restrictions

Physical site restrictions affecting disposal with or without a cap include bathymetry, currents, and water depth. The bathymetry of the potential site should be as close to horizontal as possible, since a sloping bottom could potentially affect dispersion of the bottom surge. Water currents at the site must be considered because of their effect on the descending cloud's point of impact. This effect may be calculated and accounted for, and therefore should not be a problem in the accurate placement of the dredged material. Water depth has been shown to have some effect on material placement; greater depths have a positive impact on the long-term stability of the deposited material, and the cloud of disposed material increases in size with water depth.

Practical operational requirements

Practical operational requirements include distance to the site, local obstructions, and ship traffic. These practical concerns must be addressed and evaluated for any disposal operation. Operational requirements may often be accommodated by appropriate scheduling of disposal activities.

Careful consideration of these site characteristics contribute to long-term stability of the cap and dredged material. The two types of capping include level bottom capping and confined aquatic disposal (CAD). Level bottom capping, the more common method, is the placement of dredged material and subsequent cap on flat, or nearly flat natural substrates. CAD is employed when the bottom topography requires additional supporting structures or alteration.

State of development

Cap effectiveness depends on the stability of the cap under normal tidal/wave action and storm events, and its ability to isolate underlying contaminants from the biota and water column. Capping has been extensively studied by the USACE New England Division, the U.S. Army Engineer Waterways Experiment Station (WES), and the New York District (Table 21). The following review of capping effectiveness investigations is taken from EA Engineering, Science, and Technology (1992).

A capping project in the New York Bight using dredged material from projects throughout the New York Metropolitan Area was initiated in early 1980. The disposal site was the Experimental Mud Dump (EMD) Site within the Mud Dump Site (Figure 1). Sediments from the Hudson Estuary, Newark Bay, and other waters were initially capped with fine sediments from the Bronx River and Westchester Creek and followed by a cap of sand from the Ambrose Channel (O'Connor and O'Connor 1983). The final cap was approximately 1 m thick. Numerous investigations to assess cap effectiveness were performed on the capped sediments at the EMD

Table 21
Description of DAMOS Capped Disposal Projects in the Central Long Island Sound Disposal Area[1]

Project		Materials Being Capped			Capping Method			
Date	Site Characteristics	Volume of Material yd^3	Dredging Method	Placement Method	Volume yd^3 (type of material)	Placement Method	Position Method	Thickness of Cap, ft
1979	Stamford-New Haven, North; generally flat bottom; 65 ft deep	34,000, 3 to 6 ft deep, mounded	Clamshell	Scows	65,400 (sand)	Hopper dredge	Buoy, Loran-C coupled positioning system	7 to 10
1979	Stamford-New Haven, South; generally flat bottom; 70 ft deep	50,000, 4 to 6 ft deep, mounded	Clamshell	Scows	100,000 (cohesive silt)	Scow	Buoy, Loran-C coupled positioning system	up to 13
1981	Norwalk; generally flat bottom; 65 ft deep	92,000, 8 to 12 ft deep multiple mounds	Clamshell	Scows	370,000 (silt and sand)	Scow	Buoy	up to 6 to 7
1982-1983	Mill-Quinnipiac; generally flat bottom; 65 ft deep	40,000	Clamshell	Scows	1,300,000 (silt)	Scow	Buoy	Multiple broad area placement; estimated final average of 6 to 10
1983	Cap Site No. 1; generally flat; 60 ft deep	33,000, mounded 3 ft deep	Clamshell	Scows	78,000 (silt)	Scow	Buoy	Incomplete average
1983	Cap Site No. 2; generally flat; 56 ft deep	40,000, low mound, 2 ft deep	Clamshell	Scows	40,000 (sand)	Scow	Buoy, Loran-C	Irregular, 0.6 to 4.5
1990	Long Island	34,000	Clamshell	Scows	43,000 (sand)	Hydraulic hopper dredge	Buoy	11.7

[1] Adapted from USACE (1987) and Sumeri et al. (1991).

site. Tavolaro (1984) performed a sediment budget study that quantified the dry mass of material lost in each stage from dredging to disposal. Tavolaro estimated that 5.6 percent of the overall dredged material had been lost from initiation to conclusion of the dredging project. The majority of the lost material (3.7 percent) occurred during placement at the ocean site.

Freeland et al. (1983) evaluated the short- and long-term stability of the cap placed on organic- and metal-contaminated sediments at the EMD site with field and laboratory studies. The studies determined (a) changes in the cap as a result of bottom currents and sediment transport over the winter of 1980-1981; (b) the bottom current velocities necessary to initiate erosion of the cap; and (c) the long-term probability of erosion by wind-generated waves. These studies demonstrated only slight erosion of the cap, less than the margin of error during the 1-year study period. Erosion is expected to be greater in the winter because of the unstratified water column. The low rate of erosion in the winter of 1980-1981 was attributed to a decrease in average grain size of the cap because of intermixing from other material at the EMD Site. Intermixing of grain size is believed to increase the life of the cap. Freeland et al. (1983) recognized that, although erosion was not substantial during their observations, a weakness of the cap at the EMD Site was its composition. The cap was composed primarily of fine sand that is highly erodible. Consequently, it was recommended that additional cap material of "...sand, silt, and clay consisting of mostly mineral grains with little or no organic matter, and...relatively low water content..." be added to the existing cap.

Freeland et al. (1983) also determined that a threshold current velocity of 14 to 31 cm/sec at 100 cm above the bottom was needed to initiate resuspension of the cap. Velocity data showed that mean currents at the EMD site were generally 6 to 7 cm/sec at 100 cm above the bottom, far below the threshold level. Consequently, only extreme storm events would cause the current velocity to exceed the erosional threshold. Freeland et al. (1983) found that the depth of the EMD site weakened the effect of mean surface waves; hence, as with current velocity, only storm events could produce waves great enough to impact the cap. This concern was addressed by reviewing the historical data of storm events at the EMD site. It was found that storms generating waves high enough to warrant an analysis of erosional impacts occurred only twice, 37 years apart, and therefore were not numerous enough to analyze. In summary, the investigation showed that a cap placed on deposited material in the New York Bight, specifically at the EMD site, withstands erosion impacts from wave and current action. Only infrequent storm events have the potential to create waves and currents in excess of the threshold where erosion could become a concern.

The concern of cap stability during severe storm events was addressed when hurricane Gloria passed over the capped disposal sites in the Long Island Sound (WES 1989). Erosion and sedimentation impacts from this storm were minor and confined to the immediate storm period. The primary result of this storm was resuspension of the top few centimeters of sediment at the disposal site, although bottom currents were sufficient during the storm to have caused severe erosion. Although storm events have not resulted in the erosional impacts predicted, close monitoring (and maintenance if necessary) of a cap immediately after such events is required.

Under the Disposal Area Monitoring System (DAMOS) program, capping of contaminated dredged material with cleaner sediments was investigated in several different projects. In general, these projects had favorable results. The New Haven project conducted in 1983 was considered the most successful of these. A uniformly thick cap of New Haven silt was place over Mill-Quinnipiac River chemical-containing sediment. This cap has remained stable for several years, and recolonization of natural populations of benthic infauna has begun.

To determine the effectiveness of a cap in isolating chemicals, chemical signature study was performed on the capped sediments at the EMD Site (O'Connor and Moese 1984). Vibracore samples were taken from the cap and scanned with X-rays to determine thickness of the sand cap layer. The core samples were also analyzed chemically for the compounds present in the underlying disposed sediments.

In each case, metal concentrations in the mud just below the mud-cap interface were an order of magnitude greater than the concentrations of metals in the upper layers of the sand cap. Because metal concentrations in the subcap layers remained "...virtually identical with metal concentrations found in predumping aliquots of Bronx River and Westchester Creek muds...," it was concluded that "...the uppermost layers of capped, muddy sediment had sufficient binding and sorptive surfaces to prevent even a minimum vertical flux of metals out of the muds and into the sand cap" (O'Connor and Moese 1984). As a result of the sand cap over the sediments containing metals, metal movement into the water column was eliminated and the concentration of metals available to biota reduced.

Brannon et al. (1985) examined the effectiveness of capping in chemically and biologically isolated dredged material in a laboratory and reactor units. Different capping materials were tested to determine each cap's ability to prevent dissolved oxygen depletion and ammonium-nitrogen releases into the overlying water column. A clay-sediment cap was most effective in isolating these chemicals from the water column. A silt cap was less effective than clay, and a sand cap was least effective. A cap composed of any material 50 cm thick was effective in isolating the water column in every case.

Bioaccumulation studies with sandworms (*Nereis virens*) and capping materials were also performed with contaminated sediments. Capping material with higher proportions of clay and silt was most effective in isolating burrowing worms from contaminants. However, in both studies, a 50-cm-thick cap of any material proved effective in isolating both the water column and biota from the chemicals. Thus, the thickness of a cap appears to be far more important than the composition of the cap in containing contaminants in sediment.

Chemical analyses of sandworm tissues revealed that both the 5-cm and 50-cm cap were penetrated by the worms. However, despite the increased bioturbation because of the burrowing worms, body burden levels and water column levels showed no significant increases in concentrations compared with controls of cap material only. Although bioaccumulation by burrowing worms is a concern, this problem can be addressed by investigating innate populations of benthic organisms to determine burrowing depths. These depths can then be used to determine cap thickness.

Small-scale laboratory testing of capping of New Bedford Harbor PCB-contaminated sediments by Sturgis and Gunnison (1988) indicates that a cap thickness of 35 cm effectively prevented the release of inorganic chemical tracers from the sediment into the overlying water column. A verification test measuring PCB concentrations in the water column (total PCB concentrations in the capped sediment were greater than 2,000 ppm) indicated that a capping thickness of 35 cm also prevented the movement of PCB contaminants into the water column. These tests addressed only chemical isolation of the contaminants and did not include the effects of bioturbation by burrowing organisms, nor the effects of hydrodynamic forces that may cause scoring and resuspension of cap material. Because most of the benthic organisms found in the New Bedford area do not burrow deeper than 20 cm, the thickness of cap needed to biologically and chemically isolate contaminated sediment is the sum of the chemical and the biological thickness (i.e., 55 cm) (Sturgis and Gunnison 1988). Hydrodynamic process effects require engineering considerations, and they can vary greatly from site to site.

A CAD capping demonstration was conducted on the Duwamish Waterway by the USACE Seattle District. Sediment cores taken 2 weeks and 6 months after placement of the cap showed a relatively unmixed interface between the cap and the dredged material. Analyses of lead and PCBs showed higher concentrations in the underlying dredged material, but low or below detection level concentrations in the capping material (Truitt 1986).

Capping of sediments at the Mud Dump Site in the New York Bight has been demonstrated to be a viable dredged material management technique. Capping allows environmentally sound disposal of moderately contaminated dredged material with available technology, and is thus considered a beneficial use of dredged material. With monitoring and maintenance, the structure has proven physical stability. Although bioaccumulation and the physical disruption of cap integrity by burrowing benthic organisms remains an area for further investigation, burrowing depths of local benthic populations can be determined and cap integrity can be ensured through management activities.

Pretreatment of material destined for open-water disposal is not performed generally. However, under special site-specific conditions, some conditioning of the material may be beneficial. For example, in San Francisco Bay, dredged material is passed through a screen before disposal to aid in breaking up clumps of high-clay content material. This effectively disperses material and slows the mounding of the dredged material. Conceptually, for deepwater disposal sites, the opposite conditioning, into more cohesive or higher density clumps, may be beneficial to minimize the dispersion of the discharged material in the water column. Larger, denser clumps will fall to the bottom more quickly, reducing the time ambient currents can transport material in the water column.

The effectiveness of the capping process is not affected by the method of dredging, although it is preferable to consolidate the dredged material as much as possible. It is more cost efficient to transport consolidated materials, and consolidation minimizes dispersion of the materials during disposal. A clamshell dredge used in conjunction with barge overflow operations would provide maximal consolidation of dredged materials.

The method used most often by the New York District to transport dredged materials (both contaminated materials and clean materials used for capping) is barge/scow transport. This method has proven to be reasonably cost effective and suitable for most transport activities.

Postdisposal requirements

Capping physically isolates the contaminants contained in the dredged material. There are no postdisposal requirements for controlling movement of contaminants, although confirmation of cap placement and bathymetric surveying and monitoring of the site after major storm events are required.

Feasibility of full scale implementation

The feasibility of capping has been amply demonstrated by its routine use in the New York District and New England Division of the Corps. The primary costs of capping are associated with dredging of the capping material and transporting it to the site to be capped. In some cases, the capping material must be purchased, creating another cost in addition to dredging and transportation costs. In practice, the capping material often is supplied by a separate dredging project, and costs are the same as unrestricted open-water disposal.

Capping produces no effluent. There is some near-field transient increase in suspended solids in the water column immediately after discharge of both dredged material and capping material.

No storage of dredged material or capping material is required. Both are transported directly to the site as they are dredged. No land use is associated with the capping process. Obtaining suitable capping material could conceivably involve excavation of upland soils, although all capping materials used to date have been subaqueous.

A properly placed, intact cap is effective for a wide range of contaminant concentrations. Cap thickness is of primary importance in capping effectiveness. Because capping is a nonspecific contaminant management method that is effective for all contaminants, there are no interferences with nontarget contaminants. Sediments with a range of physical characteristics can be capped, and if uncontaminated, can serve as capping materials. Hydraulically dredged muds may require special cap placement techniques because of their very poor structural ability to support a cap.

Capping requires no special site. Capping occurs directly at the dredged material disposal site. Cap stability is most easily assured at low-energy, nonerosive disposal sites, and cap erosion should be carefully considered before capping is used at other sites. Borrow pits offer lateral confinement and improved stability of the contaminated material.

The only transportation associated with the capping process is the transporting of the dredged and capping materials to the disposal site. Both are transported directly to the site as they are dredged. No specialized equipment is required. Standard dredging and transportation equipment, typically clamshell dredges and bottom-dump or split-hull barges, are used. At the disposal site, a taut-wired buoy is frequently used to mark the precise location for placement. Since the capping material is uncontaminated, no special safety precautions are required. Standard shipboard safety practices should be observed.

Regulatory requirements

No special permits are required in addition to a permit issued by the Corps of Engineers for dredged material disposal under Section 103 of the Marine Protection, Research, and Sanctuaries Act (or Section 404 of the Clean Water Act). The use of a cap to control potential impacts is sometimes specified as a condition of such permits. A state water quality certification under Section 401 of the Clean Water Act may be required for activities in state waters.

Costs

For capping of open-water disposal material, cost considerations include the same costs of unconfined open-water disposal with the additional factor of dredging, transporting, and placement of the capping material. The average costs of dredging and open-water disposal in the New York/New Jersey Harbor area is approximately $6.00 to $7.50 per cubic yard for uncontaminated sediments. This cost assumes barge overflow (to obtain the most economical loads) is permitted. For contaminated materials, overflow would probably not be allowed, and dredging transport costs are estimated to double in this case. Sediments that are defined by regulations as contaminated but still acceptable for aquatic disposal must be capped within either 2 weeks or 10 days of disposal. The time period allowed for capping is dependent upon the level of contamination; the less contaminated sediment is permitted the longer time period before capping is completed. If a sufficient quantity capping material can be supplied from other maintenance or construction dredging projects within the regulated time period, the effective costs remain the same as for unconfined open-water disposal operations.

However, effective costs increase significantly over the cost of unconfined disposal if the capping material has to be dredged and transported from a borrow area. Approximately two to three times the volume of contaminated dredged material is required to construct a cap with sufficient thickness to ensure its integrity and to isolate the underlying dredged material from the surrounding environment. All unit costs are the same as for unconfined disposal, but the increase in material to be disposed of increases the overall cost for any given volume. Average costs for dredging and capping subaquatic disposal range from $17 to $20 per cubic yard.[1] Table 22 summarizes the costs for three different volumes of dredged materials. Costs were based on an average distance between the dredging and disposal sites of 20 miles, and assumed overflow of dredged material from the barges was not permitted. The capping material was assumed to be dredged from a site only 10 miles from the disposal site, and twice the volume of contaminated dredged material was required.

Clean sediment from maintenance dredging operations can be used as the capping material. This would reduce the total costs of the two different dredging and disposal projects because the separate dredging and transport of material for capping would not be necessary. Combined unit costs for the two projects would be reduced to values similar to confined open-water disposal operations. The only constraints that make the combination of the two projects different from open-water disposal operations are that the contaminated material must be dredged and disposed first; the maintenance dredging material must be clean enough for unconfined disposal; and the disposal (capping) must occur within a short time (usually 10 or 14 days) of the contaminated material disposal.

Processing rates are assumed to be identical to unconfined disposal, although it is likely that the capping material would be collected from an unrestricted site where dredging rates could be considerably higher than production rates in areas constrained by piers, channels, and turning basins. Overall, normal dredging rates and disposal rates range from 3,000

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[1] Personal Communication, 9 December 1992, H. Hawkins, Navigation Branch Chief, New York District.

Table 22
Subaqueous Capping: Dredging and Disposal Costs

Item		Cost		
		10,000 yd^3	50,000 yd^3	100,000 yd^3
1	Dredged material			
	Dredging and transportation to disposal site[1]	$150,000	$ 680,000	$1,200,000
	Mobilization/demobilization	$ 25,000	$ 75,000	$ 100,000
2	Capping material (if dredged from borrow area)			
	Dredging and transportation to disposal site[2]	$150,000	$ 600,000	$1,200,000
	Mobilization/demobilization	$ 25,000	$ 100,000	$ 100,000
	Total Cost (1 and 2)	$350,000	$1,455,000	$2,600,000
	Unit Cost, $/yd^3$ (1 and 2)	35.00	29.10	26.00
3	Capping material (if supplied from maintenance or construction)			
	Dredging and transportation to disposal site[2]	$150,000	$ 600,000	$1,200,000
	Mobilization/demobilization	$ 25,000	$ 100,000	$ 100,000
	Total Cost (1 and 3)	$350,000	$1,455,000	$2,600,000
	Total volume removed from dredging projects, yd^3	30,000	150,000	300,000
	Unit Costs, $/yd^3$ (1 and 3)	11.67	9.70	8.67

[1] Assuming 20-mile distance and no barge overflow; if barge overflow is allowed, costs for this item would be halved.
[2] Assuming 10-mile distance and two times the volume of dredged material.

to 20,000 yd^3/day.[1] The time required to complete any one capped open-water disposal project is approximately triple that of unconfined open-water disposal. This is because the effective volume of material to be dredged and transported is doubled or tripled for capped disposal and because capping cannot begin until the completion of the disposal of the contaminated material. This brings the effective processing rate for capped disposal to between 1,000 and 7,000 yd^3/day assuming the same number of dredges and barges are used.

Assuming the capping material used is clean sediment, no land-based processes are required, and no costs associated with the lease or purchase of land are incurred. Beyond the standard dredging equipment and barges for transport, no special equipment or installations are required. Annual operating costs at an open-water disposal site would include the maintenance of marker buoys, periodic monitoring of the water quality, benthic communities, sediment quality, and the costs of surveillance and enforcement of disposal permit conditions.

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[1] Personal Communication, 11 December 1992, R. Oleiniczak, Gahagan and Bryant, Inc.

Containment Island Disposal

Description of the disposal option

Confined disposal facilities (CDFs) are a widely used alternative for dredged material disposal. CDFs are engineered structures designed to provide required storage volume for dredged material solids and to meet required effluent solid standards (i.e., to provide acceptable suspended solids concentrations for discharge to receiving waters) (USACE 1987). The principal design criterion of CDFs has been to retain as great a percentage of the fine-grained sediment particles as practical. CDFs may be located entirely upland above the water table, partially in-water, adjacent to the shore, or completely surrounded by water, i.e., the concept of a containment island (Averett et al. 1990).

Dredged material is placed in the CDF by pipeline dredges, by pumping from hopper dredges or scows, or by mechanical filling. Several site design features are common to CDFs, including containment islands. These include a perimeter retaining dike, a weir structure for release of excess water during hydraulic filling, and access for mechanical unloading. Retaining dikes are normally earth or rock fill, although slurry walls, sand, and sheet piling have been used also. Island CDFs will also require dike armoring to protect against erosion (Palermo 1988).

Within the New York and New Jersey Harbor area, upland or shoreline CDFs are not generally feasible alternatives because of the scarcity and high costs of obtaining suitable land. Material rehandling, dewatering, and transport costs would increase rapidly with distance from the dredging areas also, making remote upland sites (where land values may be lower) more expensive.

Disposal of dredged material in containment islands has been proposed as an alternative means for containing sediments. The containment islands can be centrally located and serve as regional disposal facilities. Islands can be constructed larger than is possible for shore-line or upland-based facilities and can be constructed and managed in a manner that makes them suitable for the disposal of contaminated dredged material (New York University 1989). The process of disposal into a confined facility is essentially identical to the open-water disposal option, except that dredged material is not discharged to the sea floor; it is unloaded into the CDF where it is stored permanently (Figure 23). Containment islands will have finite capacity. Once this capacity is reached, disposed sediment will then be above the water level. Capping with clean material will be required prior to converting the island to any other use.

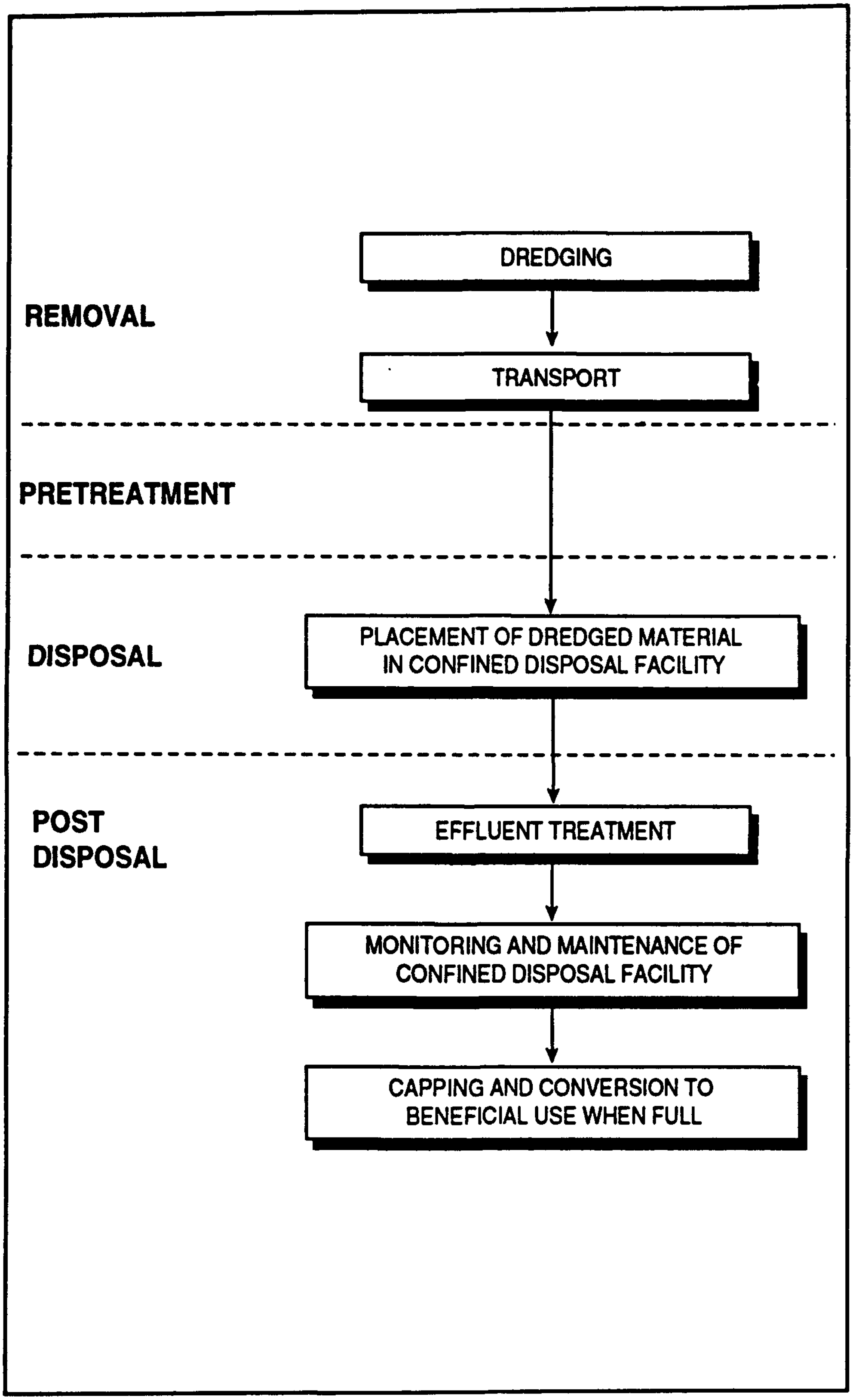

Figure 23. In-water confined disposal process

State of development

Although shore-line CDFs are in use in the Great Lakes area, development of containment islands for New York remains in the conceptual stage, although evaluations of site locations, site size, and planning costs have been undertaken (Walski and Schaefer 1988; Conner et al. 1979; USACE New York District 1980). Preliminary assessment of the in-water confined disposal option in the New York Harbor area has resulted in the identification of possible sites for containment islands (New York University 1989).

No engineering data or specifications have been developed for the design of the proposed islands. Evaluations based on annual sediment volumes of 676,000 yd^3 have been carried out in New York Harbor. These evaluations considered the engineering characteristics, the sediment characteristics, and the potential environmental impacts from chemicals and pathogens in the dredged material (New York University 1989). Preliminary cost estimates for three types of containment islands (coffer dam, sand dike, and rock dike) have been made by Walski and Schaefer (1988).

No specific predisposal processes are essential for disposal of dredged material at an island CDF, assuming the design of the facility and the associated material handling (loading/unloading) equipment is compatible with the method by which the material is dredged. That is, if both mechanical and hydraulic dredging are common, the unloading facility could be capable of both pumping material and mechanical unloading. If not mechanically dredged, material will be slurried to enable pumpout of the barge.

Maintenance of the CDF dikes, weirs, and material handling equipment will be necessary during the operational life of the facility and after it has reached capacity. Monitoring of effluent from the facility will be necessary during disposal operations, throughout the operational life of the facility, and again, after capacity has been reached and the CDF is converted to possible beneficial use, to ensure that contaminants associated with the dredged materials remain isolated from the surrounding environment.

During the operational life of the facility, effluent treatment could be required to decrease the dissolved material and suspended solids concentrations before discharge back into the surrounding waters. Varying levels of treatment are possible, from coagulation and settling to extensive filtration, and adsorption or other treatment, and disinfection.

Feasibility of full-scale implementation

This disposal option has been demonstrated and in-water confined disposal facilities, including containment islands, are presently being used to contain contaminated dredged material worldwide. Extensive use of these facilities is being made by the USACE in the Great Lakes regions. Design capacity for a 500-acre site in New York Harbor ranged from 6.6 to 28.2 million yd^3, depending upon the type of construction used for the containment site (Walski and Schaefer 1988).

Effluent control may be a necessary design component for the containment of contaminated sediments. Other design considerations would be necessary to address the possibility of contaminant loss to the environment via other pathways, such as volatilization, airborne transport of dust, and biological uptakes by plants or animals. Untreated suspended particulate matter, dissolved and particle-associated chemical contaminants, and pathogens could affect the surrounding environment. A nonanalytical evaluation of possible impacts determined that suspended solids, 16 contaminants, and several classes of pathogens would require control measures to ensure safety and compliance with water quality criteria (Poindexter et al. 1988).

Another environmental effect is destruction of benthic habitat. For a 500-acre site, 500 to 700 acres of estuarine habitat would be destroyed. This potential impact would suggest that bottom areas of low productivity be considered as possible containment island sites.

Storage of the dredged material would likely be permanent at the containment facility. Design of the facility assumes that, at capacity, the island surface will be 15 ft above sea level. Capping of the contaminated material will be necessary before the island could be converted to other industrial or recreational uses.

Contaminant concentrations and types for most New York/New Jersey Harbor sediments are not limited for containment provided that appropriate effluent treatments and controls are maintained. Sediments with contaminant levels so high that they can be classified as TSCA- or RCRA-controlled materials may be prohibited from disposal. The physical characteristics of dredged sediment will not limit the effectiveness of the containment. Hydraulic or hopper dredging has the advantage of being easier to pump into the containment island, but will result in greater volumes of effluent requiring treatment.

During construction of the facility, transportation of the dike material to the site will be required. Once operational, the only transportation required will be the barging of dredged material for confinement.

Regulatory requirements

An EIS will be necessary as part of the implementation of an in-water CDF. Public concerns of an island CDF would be addressed during the EIS procedure. Comprehensive permitting requirements would be developed by regulatory agencies to control the disposal at the facility and also control the effluent quality discharged from the facility.

Costs

Planning-level cost estimates for the construction of a 500-acre dredged material containment island in New York Harbor are based on the estimates of Walski and Schaefer (1988) and are adjusted for 1992 dollars (Table 23). These estimates are "order-of-magnitude" values. Islands with sand dike, rock dike, and cofferdam construction were considered. Because of the wide range of effluent treatment costs and the uncertainty regarding the required level of treatment, an average increase of 5 percent was applied to the unit costs of construction to cover the cost of postdisposal effluent treatment. Unloading of dredged material from barges and placement in the CDF is assumed to be performed by hydraulic pumps. Unloading costs are based on Walski and Schafer's (1988) estimates for dredging and placing sand for the construction of dikes. Adjusted to 1992 dollars, these estimates range from $2.80 to $5.60 per cubic yard. A middle value of $4.00 per cubic yard was used to determine the cost estimates for this disposal alternative.

To make the costs comparable to other disposal options and treatment options, the cost of removal and transport of material from the dredging site to the in-water CDF were included. The same three volumes of material (10,000, 50,000, and 100,000 yd^3) were used as capacities for this CDF option, although a much larger capacity facility would be more practical and have a longer operational life. Unit costs for disposal at sand or rock dike islands ranged between $39 and $40 per cubic yard. Costs for the sheet pile cofferdam construction and disposal operations were estimated to be between $24 and $32 per cubic yard. The variation in construction costs (dredging and disposal operation costs varied with volume only) reflects the high cost of obtaining and transporting the sand or rock material to a proposed confinement island site.

Unit costs for the sheet-pile cofferdam option are similar to the costs associated with the open-water capping option when the capping material is dredged specifically for that purpose. If the capping material can be supplied as a result of a separate dredging project, unit costs for capping approximate those of unrestricted open-water disposal; then the confined disposal island alternative becomes approximately three times more expensive.

Table 23
Cost Analysis: Island Confined Disposal Facilities

Construction Type	Type of Cost	Cost ($,000s) for		
		10,000 yd^3	50,000 yd^3	100,000 yd^3
Sand Dike				
Dredging and transportation[1]	Variable	175	755	1,300
Unloading barges at CDF	Fixed	40	200	400
Construction of island CDF[2]	Variable	347	1,735	3,472
Total		562	2,691	5,072
Cost, $/$yd^3$		56.20	53.82	50.72
Rock Dike				
Dredging and transportation[1]	Variable	175	755	1,300
Unloading barges at CDF	Fixed	40	200	400
Construction of island CDF[2]	Variable	264	1,319	2,638
Total		479	2,274	4,338
Cost, $/$yd^3$		47.90	45.48	43.38
Sheet-Pile Cofferdam				
Dredging and transportation[1]	Variable	175	775	1,300
Unloading barges at CDF	Fixed	40	200	400
Construction of island CDF[2]	Variable	121	607	1,214
Total		336	1,582	2,914
Cost, $/$yd^3$		33.60	31.64	29.14

Note: All values are in 1992 dollars (Walski and Schaefer 1988).

[1] Includes mobilization/demobilization costs (see Table 22).

[2] Includes design, construction, maintenance, and effluent treatment costs.

5 Evaluation of Treatment and Disposal Alternatives

Bench-Scale Test Results

Bench-scale tests of four of the six treatment technologies described in the previous chapter have been performed. These tests were carried out by or in conjunction with the developers of the particular technologies using representative small samples of dioxin-contaminated sediments collected from New York/New Jersey Harbor. These samples were split from a 60-gal composite sample collected in July 1992 from a location known to be contaminated with dioxins. The four treatments are as follows:

- Thermal gas-phase reduction.
- Solvent extraction.
- Base-catalyzed dechlorination.
- Incineration.

Thermal gas-phase reduction

Three treatability test runs were performed with three sediment samples of approximately 5 gal each. Each laboratory-scale test involved pretreating the samples in a separate thermal desorption unit (600 °C) before injection into the thermal (900 °C) gas-phase reduction reactor (Eco Logic International, Inc. 1992b).

The samples were pretreated by screening through a 1/8-in. mesh and mixing by hand. Each test run lasted from 3 to 4 hr with a 2-min residence time in the thermal desorption unit. The evaluation of the process destruction efficiency calculation was based on the results of chemical analyses for dioxins and furans in the sample material before treatment and in the treated processed solids. Analyses were performed by the developer (Eco Logic International, Inc. 1992b) for various PCDDs (the analytical methods are not specified) and independently by IT Corporation Analytical Services laboratory (ITAS) for the specific isomer 2,3,7,8-TCDD and total TCDD as well as other isomers and total PCDDs using a modified USEPA Method 8290 (IT Corporation 1993a). Results vary significantly between the two laboratories, the independent IT laboratory reporting higher concentrations in both feed and product material. These results are summarized in Table 24. Eco Logic reported that the low decontamination efficiencies demonstrated for their process were due to a thermal desorption residence time that was too short. Eco Logic suggests that extending the 2-min residence time used for this test to 10 min will greatly improve the efficiency. Eco Logic reported destruction efficiencies of polychlorinated dibenzo dioxins concentrations in the gas-phase reduction reactor as greater than 95 percent.

Table 24
Results from Bench-Scale Tests of Dioxin-Contaminated Sediments

	2,3,7,8-TCDD, pptr			Total TCDD, pptr		
Process	Feed	Product	Efficiency[3] %	Feed	Product	Efficiency %
Gas-Phase Thermal Reduction	159 130[1]	99.7 50[1]	37.3 61.5	193 130[1]	144 50[1]	25.4 61.5
Solvent Extraction (B.E.S.T.)	119 217[1]	16.5 15[1]	86.1 93.1	136 276[1]	21.3 15[1]	84.3 94.6
Base-Catalyzed Dechlorination (1)	268[4] 67.5[1]	< 1.9[2] < 0.996[1]	>99.3 >98.5	309[4] 83.7[1]	< 11.1 < 0.301[1]	>96.4 >99.6
Base-Catalyzed Dechlorination (2)	268 67.5[1]	<33.1 <11.0[1]	>87.7 >83.7	309 83.7[1]	< 47.8 < 11.0[1]	>84.5 >86.9
Incineration	289	< 0.73[2,4]	>99.8	255	< 1.4[2,4]	>99.5

[1] Analyses performed by technology developer.
[2] Not detected, reported as analysis detection limit.
[3] % Efficiency = (Feed conc. - product conc.) x 100/Feed conc.
[4] Average of two samples.

Solvent extraction

Bench-scale treatability tests were performed on one sediment sample using the B.E.S.T. process developed by Resource Conservation Company. Pretreatment consisted of blending the sample and screening to 1/4-in. mesh. A compatibility test of the sample with triethylamine was performed, and the pH of the sample was adjusted to a value of 11 by the addition of sodium hydroxide. A total of six extraction stages were completed; after each stage, the solids were returned to the extraction vessel prior to the following extraction (Resource Conservation Company 1993).

Samples of the untreated sediment and the treated product were analyzed at both a developer-specified laboratory (ENSECO) and at ITAS using methods described in USEPA SW-846, Method 8290 (IT Corporation 1993b). Removal efficiencies calculated from analyses from the two separate laboratories agreed to within 10 percent (Table 24).

Base-catalyzed dechlorination

Sediment samples were treated at Wright State University, OH, in coordination with USEPA's RREL. A small capacity laboratory-scale reactor was used for the tests. Two tests were performed and results are reported in Table 24. For the first test, 50 g of sediment were mixed with water, sodium hydroxide, tetraethylene glycol, and a proprietary catalyst. The mixture was heated to 320 to 345 °C for a period of 90 min. In the second test, sediment was mixed with sodium hydroxide, a high-boiling point hydrocarbon oil (LW-110), a solid catalyst, and a liquid catalyst, both proprietary to the USEPA. The mixture was heated to 320 to 345 °C for a period of 4 hr. In both cases, the dioxins in the treated material were below the detection limit. Although dioxin destruction efficiencies appear to be less for the second test because of a higher detection limit, the conditions used for the second test were much more effective in also destroying the PCBs in the sample (from a feed concentration of 281 ppb to a product concentration of <2 ppb) (Tiernan 1993).

Incineration

A small-scale thermal test unit, installed at USEPA's IRF and operated by Acurex Environmental Corporation, was used to treat 27 lb of sediment. The test program consisted of five tests at two charge chamber incineration temperatures (815 and 980 °C) and two incineration residence times of 0.5 and 1.0 hr (Acurex Environmental Corporation 1992; Siag, Venkatesh, and Waterland 1992).

Samples of the treated sediment (one sample for each of the five tests) and one sample of the untreated (feed) sediment were sent to ITAS for dioxin and furan analyses. Reduction efficiencies were calculated from the results of these analyses, based on the reduction in concentration of the contaminants between the untreated and treated solids. Because many analyses reported below detection limit concentrations, minimum efficiencies were calculated based on the detection limit value as the upper bound of the concentration. All TCDD concentrations in the treated sediments were reported at below detection limit. Efficiencies were calculated for five tests and for the same two compounds reported in the previously discussed treatments. These values are summarized in Table 24. The complete data sets are available.[1]

Because this is an incineration process, DREs can be calculated by combining sediment feed rates and flue gas discharge flow rates with the reported sediment and flue gas concentrations. The highest percentages reported for any one test were 99.989 for total TCDD, and >99.994 for 2,3,7,8-TCDD. No DREs were greater than the 99.9999 percent required for the incineration of dioxin-contaminated hazardous waste. The investigator indicated that this was a result of low initial dioxin concentrations in the untreated sediment and that the measured flue gas concentrations were typical for the incineration of any wastes containing chlorine compounds (Waterland 1993).[1]

Table 24 presents the preliminary results for the treatability studies of the four treatments in terms of the reduction of the individual isomer 2,3,7,8-TCDD and the total TCDD concentrations in the samples. These values are based on chemical analyses of untreated sediment samples and the solid residual after treatment. Results are reported as the concentrations measured or as less than the analytical method detection limit in the cases where no dioxin congeners or total dioxin were detected. Consequently, contaminant reduction efficiencies calculated from the nondetectable values are minimum efficiencies only, and actual efficiencies may be greater.

Two processes, incineration and base-catalyzed dechlorination, were reported to have nondetectable levels of dioxin in the treated product. Total TCDD reductions greater than 96 percent were attained for both treatments. Reduction efficiencies for the 2,3,7,8-TCDD congener were slightly higher, at 99.3 and 99.8 percent, respectively.

Results from the other two processes, solvent extraction and gas-phase thermal reduction, show that TCDD congeners were still detectable in the sediment after treatment. Between 84 and 86 percent of 2,3,7,8-TCDD and total TCDD were removed by the extraction process. The feed material and treated product from the gas-phase thermal reduction process were analyzed by both the vendor and an independent laboratory. The results of the two analyses differed greatly. Pretreatment and posttreatment dioxin concentrations reported by the vendor were approximately one-half, and the calculated reduction efficiency (61.5 percent) approximately doubles the corresponding values measured by the independent laboratory (37.3 percent efficiency for 2,3,7,8-TCDD and 25.4 percent for total TCDD).

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[1] Personal Communication, 16 March 1993, L. Waterland, Acurex Environmental Corporation, Mountain View, CA, Letter to D. Averett, WES, Vicksburg, MS.

These preliminary bench-scale test results suggest that incineration and base-catalyzed dechlorination processes are the most efficient at destroying or removing dioxins, achieving at least a 96-percent TCDD reduction efficiency. The solvent extraction process is the next most efficient, removing between 84 and 86 percent, and the thermal reduction process the least efficient, reducing TCDD concentrations in the sediment by 25 to 61 percent.

Criteria for Comparisons

Several previous studies have reported on developed technology and process screening methodologies for treatment alternatives for the cleanup of contaminated soils, sludges, and sediments (Averett et al. 1990; Cullinane et al. 1986; USEPA 1988). This assessment follows the approach developed in USEPA (1988) and adapted by Averett et al. (1990). Three broad criteria are used to produce a relative ranking of the six process options described in Chapter 4. This is a screening level evaluation only. A much wider range of factors would be investigated to determine which processes would be pursued for a more detailed evaluation. These criteria are as follows:

- Effectiveness in destroying or removing dioxins.
- Implementability.
- Cost.

Effectiveness

A complete evaluation of effectiveness involves (a) the capability of the alternative to meet cleanup objectives, (b) impacts to human health and the environment during construction and implementation, and (c) the demonstrated performance and reliability of the process. This analysis relies primarily on part (a) since the other factors require a more subjective rating and they were not reviewed in depth for this preliminary screening of alternatives. For this evaluation, effectiveness is derived directly from the dioxin-contamination reduction efficiency of the process, which is defined as that fraction of the contaminant in the feedstock that is destroyed or removed by the process. The preliminary results from the bench-scale testing are used as the basis of this ranking. For the two treatment alternatives not tested, bioremediation and solidification, little information is available on the actual efficiency of either technology for reducing or stabilizing dioxin-contaminant concentrations. Estimates of efficiency of less than 50 percent, based on the available literature and the treatment of either dioxins or PCBs, are used.

Only the dioxin reduction efficiency was considered in this ranking. The reduction efficiency of other contaminants known to exist in New York/New Jersey Harbor sediments and found in conjunction with dioxins, such as heavy metals, is not addressed. The following ranking is applied to this factor:

Ranking = 3	The process option is capable of achieving a dioxin-contamination reduction efficiency of greater than 99 percent.
Ranking = 2	The process option has been shown to achieve a dioxin-contamination reduction efficiency of between 50 and 99 percent.
Ranking = 1	The process option has not been demonstrated to achieve a dioxin-contamination reduction efficiency of better than 50 percent.

Implementability

This criterion covers many aspects of developing the process for commercial application and of the suitability of the process to the reduction of dioxin contamination. These factors are discussed in the technology descriptions presented in Chapter 3. They include availability, process limitations, pretreatment and posttreatment requirements, processing rates, time required to construct a commercial-sized unit, and special site requirements. The following ranking is necessarily more subjective than the other evaluation factors because of the limited development of commercial treatment units for dioxin-contaminated materials.

Ranking = 3	The process option is commercially available; it has proven applicability to dioxin-contamination soils and sediments; and it has been demonstrated at field or commercial scales.
Ranking = 2	The process option is commercially available, and it has been demonstrated at bench scales for dioxin-contaminated soils and sediments.
Ranking = 1	The process option has been demonstrated at bench or laboratory scales to be applicable to dioxin-contaminated soils and sediments or is an emerging technology that requires additional development and testing.

Costs

The cost ranking is developed from the individual technology cost tables presented in Chapters 3 and 4. Costs are calculated on a present value basis and include reported or estimated costs associated with the complete treatment alternative. This includes all the appropriate processes from the component technologies (removal, transport, pretreatment, treatment, posttreatment, and disposal). Unit costs were provided for three different volumes of material to be treated, 10,000, 50,000, and 100,000 yd^3. A summary of the estimated unit costs, based on the information in the cost tables, is presented in Figures 24 and 25.

The error bars shown in the figures indicate the estimated accuracy of the predicted costs. For the treatment alternatives, the range is estimated to be between -60 and +100 percent. For the disposal options, the estimated accuracies vary. Unrestricted ocean disposal costs and capping costs may vary by about 20 percent. Containment island costs are derived from order-of-magnitude estimates (Walski and Schaefer 1988) and, thus, less precise than the other two options. In this case, the error bars extend beyond the limits of the graph. Scaling the graph to include these error bars would obliterate the variations in costs of the other two options.

This ranking uses a single value for the unit cost—the estimated cost for treating 50,000 yd^3 as the representative cost for each process option. This value is approximately equal to the average of the unit costs calculated for each of the three volumes considered.

Ranking = 3	Unit cost (dollars/cubic yard) for the treatment alternative is less than $300.
Ranking = 2	Unit cost (dollars/cubic yard) for the treatment alternative is between $300 and $600.
Ranking = 1	Unit cost (dollars/cubic yard) for the treatment alternative is greater than $600.

Conclusions and Recommendations

Treatment options

Results of the comparative evaluation, as demonstrated by the composite score in Table 25, indicate that the treatment alternatives can be classified into two groups. One group of three processes (thermal gas-phase reduction, bioremediation, and solidification) each received a relative low score of 5; the other group of three (solvent extraction, incineration, and base-catalyzed dechlorination) each received a relative high score of 7 out of a total of 9. The three with the lower score each scored low on

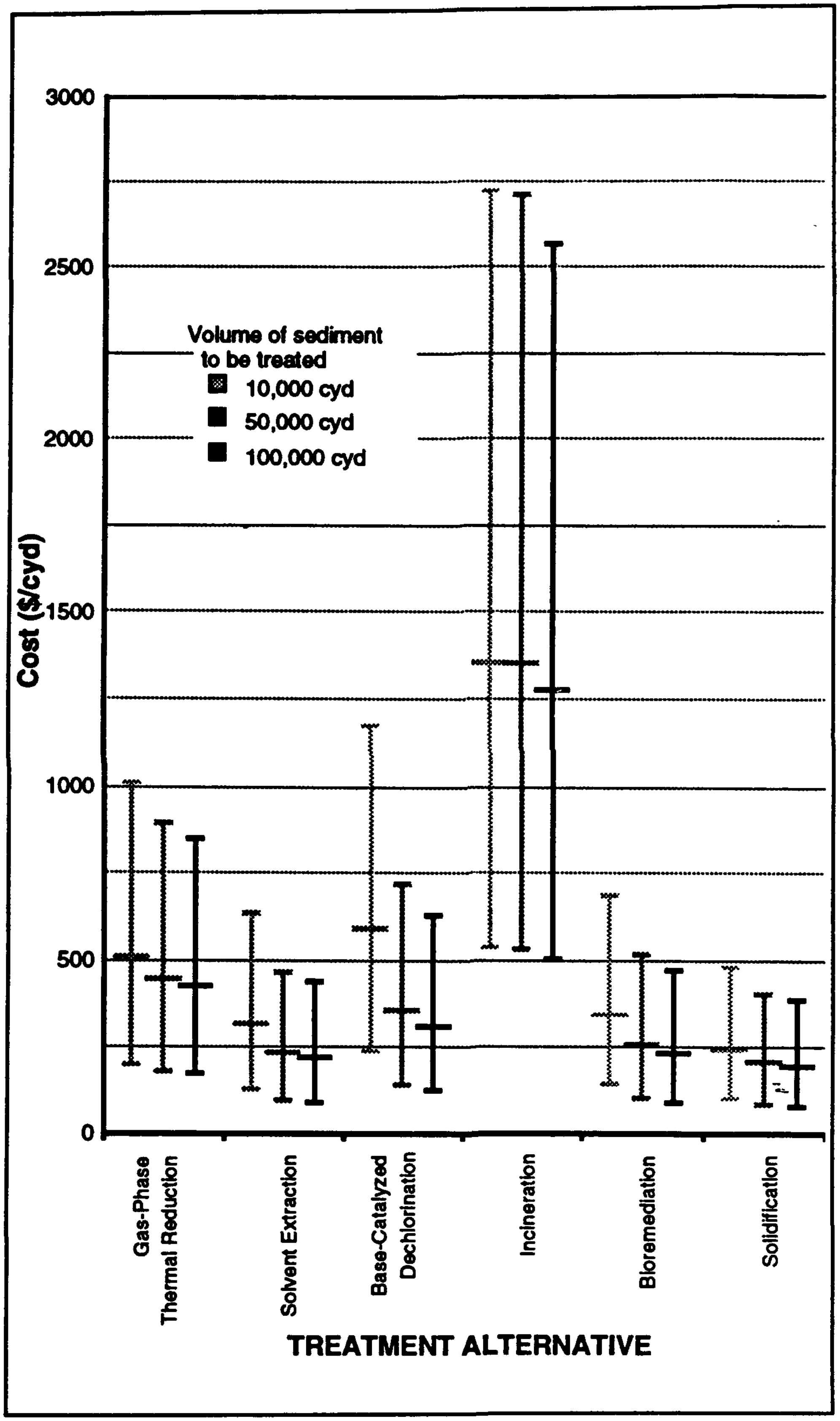

Figure 24. Estimated unit costs for treatment alternatives

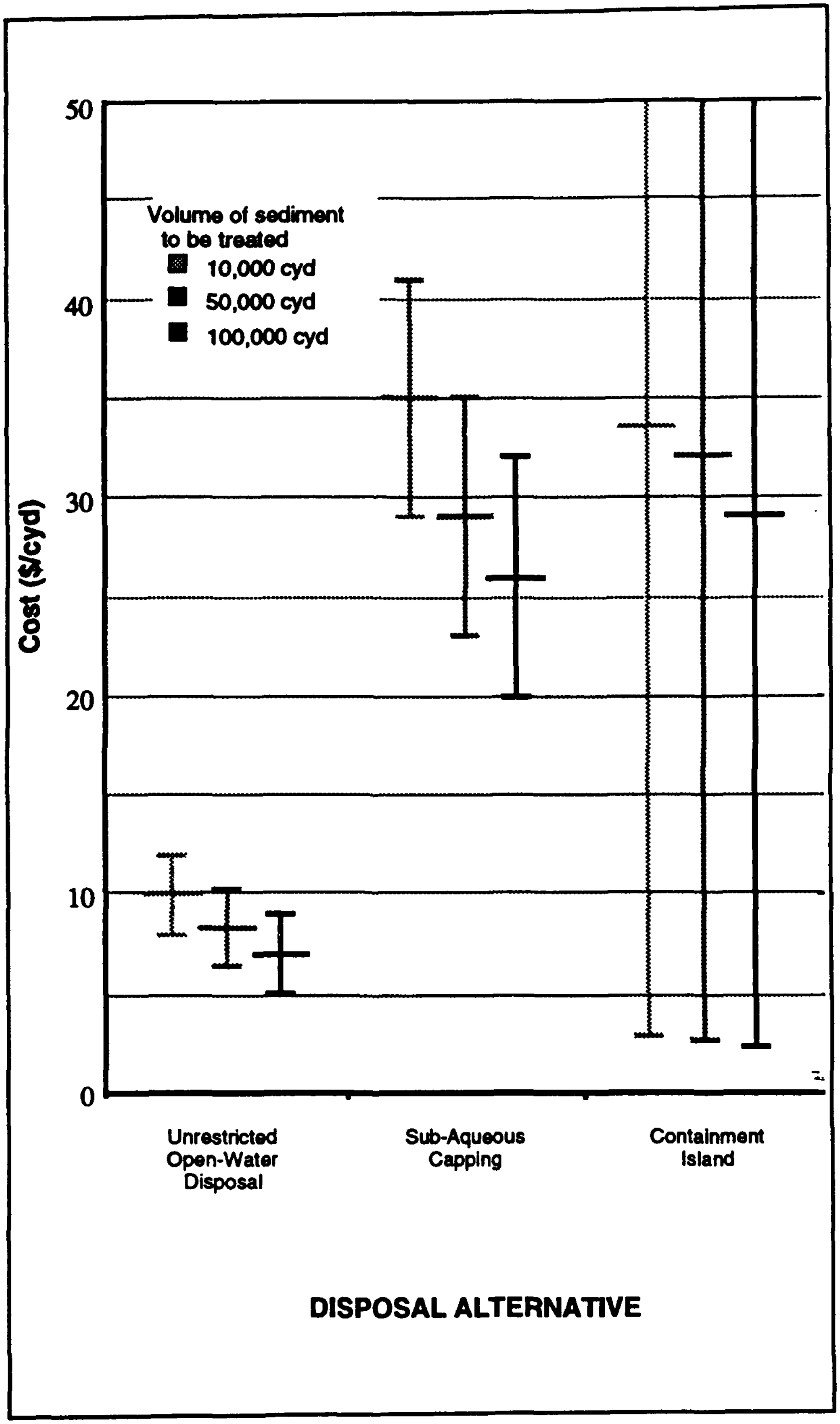

Figure 25. Estimated unit costs for disposal alternatives

effectiveness and implementability, although the scores for cost were high. The three processes scoring higher did so in general because of high effectiveness and medium to high implementability. The cost score for these three processes ranged from low to high.

Table 25
Comparative Evaluation of Treatment and Disposal Alternatives

Treatment Alternative	Evaluation Factors Effectiveness	Implementability	Cost	Composite Score
Thermal Gas-Phase Reduction	1	2	2	5
Solvent Extraction	2	2	3	7
Incineration	3	3	1	7
Base-Catalyzed Dechlorination	3	2	2	7
Bioremediation	1	1	3	5
Solidification	1	2	3	6
Disposal Alternative				
Open-Water Disposal	1	1	3	5
Subaqueous Capping	2	3	3	8
Island Confined Disposal	3	3	3	9

Incineration has proven to be highly effective in bench-scale tests and to have demonstrated commercial application to dioxin-contaminated sediments, soils, and sludges; but it is also the most expensive, approximately four times greater than any other option.

Solvent extraction scored in the midrange for effectiveness. However, effectiveness might be improved by increasing the number of extraction stages in the treatment. Implementability also scored in the midrange because the process, although commercially available, has not yet been demonstrated with dioxin contamination at this scale. The cost score was high, a result of unit costs of under $300 per cubic yard.

Base-catalyzed dechlorination was shown to be highly effective in the bench-scale tests. The implementability score was midrange because the process has not yet been demonstrated on large quantities of dioxin-contaminated material, although the treatment equipment is commercially available. Costs reported in Chapter 3 were estimated from the limited application of the process with PCBs. They fell within the midrange also.

Containment disposal options

The evaluation factors discussed previously and used with the treatment options require a slightly different interpretation when applied to disposal options. Because the material is not treated in any way, contaminant concentrations are not reduced. Effectiveness then becomes a measure of how well the process prevents the target contaminant from entering the environment. Implementability of the disposal options becomes less dependent on the physical processes and more dependent on other factors, such as regulatory control, long-term effectiveness, and public acceptance. Disposal option costs are between one and two orders of magnitude less than the treatment options, so all disposal options score the highest value for this factor (Table 25).

Open-water disposal received the lowest composite score of 5. This resulted from lower scores for effectiveness and implementability. This disposal method does little to control the migration of contamination into the environment at the disposal site. Implementability is reduced because of regulatory prohibitions against open-water disposal of highly contaminated sediments.

Subaqueous capping of disposed material with clean dredged material or fill has been shown to limit the long-term environmental impacts of contamination by isolating the contaminated material from erosion and bioturbation effects. Capping also preserves the geohydrologic and geochemical conditions in the disposed material, minimizing the release or migration of contaminants into the surrounding water column or sediments (Averett et al. 1990). However, specific guidelines, based on bioaccumulation criteria, that limit dioxin concentrations have been proposed for the disposal of dioxin-contaminated material in New York Bight (USACE New York District 1992). Thus, implementability scores are higher than for open-water disposal, but the option is still subject to significant contaminant concentration constraints.

In-water confined disposal received the highest scores for all evaluation factors. This is a demonstrated and effective method of controlling the entry of contaminants into the environment surrounding the disposal site. Although unit costs are two to five times greater than for the other disposal options (Figure 25), they are still at least 10 times less than the lowest-cost treatment option (Figure 24).

Recommendations for the future

The desktop and laboratory evaluations performed by this study to evaluate treatment options for NY/NJ Harbor sediments indicate that treatment of NY/NJ sediment is technically feasible. Although target cleanup levels for 2,3,7,8-TCDD have yet to be established, the best available technology operating at very high efficiency will likely be required. Two of the technologies evaluated by bench-scale studies approached treatment levels of one pptr of 2,3,7,8-TCDD in the laboratory. Perhaps the other two technologies evaluated as well as other processes can approach this level with additional optimization of process conditions. However, most treatment processes prove to be somewhat less effective when scaled up to pilot or full-scale. Pilot-scale evaluations also demonstrate materials handling and mass transport limitations that become evident in dealing with volumes of material more representative of full-scale operating conditions. Pilot-scale evaluations also significantly improve the capability to estimate full-scale costs.

Future evaluations of cleanup options for NY/NJ Harbor sediments should include a pilot-scale evaluation of one or more treatment technologies. If processes other than the four evaluated in the laboratory for this study are chosen, the additional bench-scale tests should be performed prior to proceeding to the pilot evaluation. Pilot-scale data will benefit the assessment of the efficacy of a full-scale cleanup of contaminated sediment sites in NY/NJ Harbor.

While future evaluations of treatment technologies on a laboratory or pilot scale will allow better decision making for selecting and designing a cleanup project, they will not significantly alter the fact that treatment would likely increase the cost of sediment disposal by an order of magnitude or more. Innovations in treatment technology and large-scale projects will reduce the difference in cost, but only to a limited degree. If funding were available to design and install a fixed plant that would treat NY/NJ Harbor sediment on a continuous basis for a number of years, the cost may be reduced considerably. A cleanup project on this scale would be unprecedented, and the capital required to implement such a project would require major government funding.

Other obstacles to implementing large-scale treatment projects are the shortage of available real estate, permitting and regulatory issues, environmental and cultural concerns, disposal of treated residues and side streams, financing, public acceptance, and most of all, economics. It would need to be justified for the purpose, whether it be to maintain or construct a navigation area or to clean up the environment. Because of the expense and difficulty in implementing a sediment cleanup project for this area, plans for implementation of cleanup projects should focus on those projects that provide the greatest environmental benefits. Generally, these projects are those with higher contaminant concentrations, where lack of action will allow for further dispersion of the contamination, and where capped open-water disposal cannot be implemented.

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The U.S. Army Engineer District, New York, dredges several million cubic yards of sediment annually to maintain Federal navigation channels in New York and New Jersey Harbor. Most of this dredged material has characteristics that allow its disposal into open water or ocean sites. However, contaminant concentrations in some of the materials have led to this and other investigations of alternate management techniques for dredged material that is unacceptable for open-water disposal. These alternatives include ocean disposal with capping, coastal borrow pit disposal with capping, land-based or in-water confined disposal, and treatment of sediment to reduce the contaminant concentrations to levels acceptable for unrestricted disposal or beneficial uses.

This report assesses available treatment and disposal alternatives for dioxin-contaminated dredged material from New York/New Jersey Harbor. Included in the assessment of treatment alternatives are a survey of available options, results from bench-scale tests of selected treatment technologies, development of the overall process train for promising treatment alternatives, an assessment of the feasibility of implementing the alternative, preliminary cost estimates, and a comparison of alternatives. Disposal alternatives are discussed on a similar basis and are compared with treatment alternatives.

14. SUBJECT TERMS
Bioremediation
Capping
Confined disposal
Contaminated
Dechlorination
Dioxins
Disposal
Dredged material
Dredging
Extraction
Incineration
Remediation
Sediment
Solidification
Thermal
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