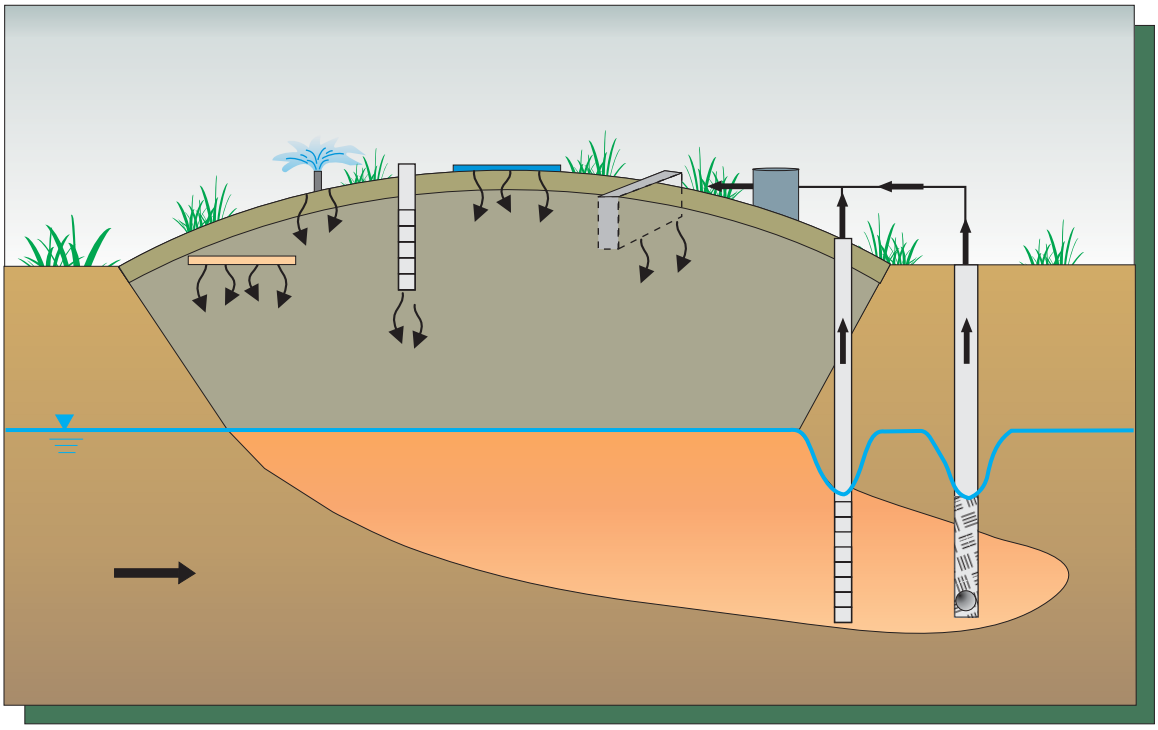
FINAL

Impact of Landfill Closure Designs on Long-Term Natural Attenuation of Chlorinated Hydrocarbons



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FINAL

IMPACT OF LANDFILL CLOSURE DESIGNS ON LONG-TERM NATURAL ATTENUATION OF CHLORINATED HYDROCARBONS

March 2002

Prepared for:

Environmental Security Technology Certification Program
Arlington, Virginia

CONTRACT NO. DACA72-00-C-0013

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EXECUTIVE SUMMARY

The purpose of this landfill-closure evaluation project is to assist the Environmental Security Technology Certification Program (ESTCP) and their United States (US) Department of Defense (DoD) technology-transition partners in developing alternative landfill closure designs and management strategies that can enhance the long-term natural attenuation of chlorinated aliphatic hydrocarbons (CAHs) (i.e., chlorinated solvents) in landfills and landfill-leachate-contaminated groundwater. The project was divided into three primary tasks:

- **Task 1 – Literature Review/Landfill Data Review**. This task involved completion of a literature review that focused on determining how alternative landfill closure designs and landfill management strategies can impact the natural attenuation of and long-term risk associated with CAH groundwater plumes.

- **Task 2 - Conceptual Landfill-Design Model**. This task involved preparation of simple decision-logic diagrams (Initial Landfill Screening, Remedial Alternatives Evaluation, and Cover Type Selection) and several design alternatives that could be used to optimize CAH biodegradation while minimizing plume migration.

- **Task 3- Final Technical Report.** This technical report describes the decision-logic diagrams and is intended to provide DoD engineers with a useful tool for evaluating the potential for enhancing CAH biodegradation at landfills undergoing final closure planning and design.

ES.1 BACKGROUND AND POTENTIAL BENEFITS

The 'enhanced-leaching' theory of landfill treatment challenges the conventional philosophy of 'infinite containment,' and could move the waste industry toward more permanent and efficient waste-management solutions. These solutions could include a greater reliance on evapotranspiration (ET) landfill covers using natural vegetation, and greater emphasis on natural attenuation rather than expensive leachate collection and treatment systems.

At some landfills, surface infiltration may accelerate the leaching of the "source" and reduce the time required for biological stabilization of the landfilled waste. Recirculation of landfill leachate could also be used to accelerate the source-leaching process and promote reductive dechlorination within a "closed loop" *in-situ* bioreactor. This could reduce the time required for a CAH plume to stabilize and degrade, thereby reducing the long-term risk posed by the site. More rapid waste stabilization also will reduce long-term monitoring (LTM) costs because most regulatory agencies require rigorous post-closure monitoring activities as long as wastes pose a potential threat to water quality.

ES.2 REGULATORY ENVIRONMENT

In 1993, the US Environmental Protection Agency (USEPA) established source containment as the presumptive remedy for municipal solid waste (MSW) landfills regulated under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA). In 1996, USEPA stated that this presumptive remedy, including installation of a landfill cap to minimize infiltration, should also be applied to all appropriate military landfills. Therefore, the concept of facilitating *in situ* degradation of landfilled wastes is at odds with the implicit intent of waste isolation and infiltration minimization. Implementation of an alternative landfill management strategy would require profound changes in the current regulatory paradigm.

However, recent actions by USEPA indicate that the regulatory climate may be changing, thus making regulatory approval of more innovative MSW landfill closure strategies more likely. For example, in recent years USEPA has been focusing on risk-based/performance-based corrective action approaches that encourage technical and administrative innovation to achieve environmentally protective cleanups on a facility-specific basis. The Resource Conservation and Recovery Act (RCRA) Reforms of 2001 promote use of innovative approaches, accelerate changes in regulatory culture, and focus on remediating to capitalize on long-term development potential. These paradigm shifts appear to create a more favorable framework within which to explore innovative management strategies for MSW landfills that are intended to degrade and/or minimize the potential long-term threat of landfilled wastes.

ES.3 INITIAL LANDFILL SCREENING

A Landfill Screening Decision Tree was developed to assist Department of Defense remedial program managers in determining if engineered remediation of CAHs in groundwater is required, or if cleanup goals can be met within an acceptable time frame via natural attenuation. The initial screening process involves determining the presence/absence of contamination, and (if significant contamination is present) performing a natural-attenuation treatability study (TS), contaminant fate and transport analysis, and risk analysis.

If site assessment results indicate that significant contamination is not present, then it is unlikely that a final cap consisting of a low-permeability cover is required. If contaminant fate and transport analyses indicate the potential for the CAH plume to migrate off government-controlled property, and downgradient receptors are potentially at risk, then a detailed remedial alternatives assessment should be completed, using the decision tree developed herein for this purpose, to select an appropriate remedy that will mitigate risks posed by the plume. Conversely, if cleanup goals can be met within an acceptable time frame via natural attenuation, then a remedial alternative consisting of monitored natural attenuation (MNA) of groundwater should be pursued.

ES.4 REMEDIAL ALTERNATIVES ASSESSMENT

The Remedial Alternatives Decision Tree guides the user in selecting an appropriate remedial alternative when risk or time considerations dictate a more aggressive approach than MNA alone, and indicates the conditions under which operation of the landfill as a

bioreactor, with collection and recirculation of leachate-contaminated groundwater, is an appropriate alternative. Elements of this decision tree include:

- Source Identification and Removal: If an active, continuing source likely is present, then reasonable efforts to locate and remove/remediate significant sources may provide a significant payback in terms of reduced plume migration and persistence.

- Determining the Feasibility of Plume Capture and Recirculation: Site-specific hydraulic models can be developed to design plume capture systems and determine optimal leachate recirculation rates. If the dissolved CAH plume can be contained by extracting flow from the impacted aquifer, which can then be reapplied within the landfill without adversely affecting the site, then recirculation of leachate-contaminated groundwater and operation of the landfill as a bioreactor should be considered. This *in situ* bioremediation alternative can potentially accelerate the waste-stabilization and contaminant-mass-reduction processes at problematic landfills for a relatively low cost. For older landfills such as those maintained by the DoD, leachate recirculation may enhance degradation of persistent chemicals by providing more favorable conditions for the microorganisms of interest. If recirculation of leachate-contaminated groundwater or MNA is not a feasible remedial alternative, then use of a permeable reactive barrier and/or enhanced bioremediation of dissolved CAHs should be considered. Enhanced bioremediation can include creation of anaerobic zones for tetrachloroethene/trichloroethene (PCE/TCE) degradation, and/or creation of aerobic zones for dichloroethene/vinyl chloride (DCE/VC) degradation.

- Bioreactor Options: Bioreactor landfills can conceivably be managed to promote either anaerobic conditions (for PCE and TCE) or aerobic conditions (for DCE and VC), or sequential zones of both conditions. Aerobic conditions can be created or maintained by recirculation of naturally aerobic or mechanically aerated groundwater, or via addition of oxygen or an oxygen source into the subsurface. Anaerobic conditions can be created or maintained by recirculation of anaerobic, organic-rich groundwater, with or without amendments such as electron donors. Anaerobic microbial degradation processes that can be artificially enhanced, depending on site-specific conditions, include direct and cometabolic reductive dechlorination, and direct anaerobic oxidation. Aerobic processes that can be enhanced include direct and cometabolic oxidation. Creation or enhancement of sequential anaerobic and aerobic treatment zones represents the most promising scenario for successful biodegradation of the full range of CAHs that may be present at landfill sites. In addition, pH adjustment or addition of nutrients may be desirable to maintain a healthy microbial population. Successful implementation of leachate recirculation depends on:

- Collecting and controlling leachate as it is generated;

- Effectively redistributing leachate throughout the landfill without creating leachate seeps or other exposure-related issues;

- Controlling recirculation in a fashion that promotes robust microbial activity; and

- Containing and collecting gases produced as a result of biodegradation (typically not of concern at most DoD landfills due to their age and relatively stabilized nature).

ES.5 LANDFILL COVER DESIGNS

Three general types of covers are recommended for consideration at DoD landfills, depending on site-specific characteristics: low-permeability covers, ET covers, and capillary-barrier covers. In some cases, maintenance or enhancement of the existing soil cover may be adequate to achieve remedial goals. The experimental nature of capillary barriers, and mixed results from prior applications of this cover type, suggest that they should not be used alone, but could be incorporated into an ET cover design.

Low-permeability covers should be considered if the contaminant source material generally remains above the seasonally high water table, and/or the climate is characterized by abundant rainfall that would overwhelm the capacity of an alternative cover to adequately control infiltration. In all other cases, alternative (e.g., ET) covers should be considered when it is desirable to eliminate or reduce infiltration of water. If highly chlorinated solvents such as PCE and TCE are the primary risk drivers in the groundwater plume, an organic (e.g., mulch) layer can by incorporated into an alternative landfill cover to scavenge available oxygen and enhance anaerobic biodegradation processes within and beneath the landfill.

ES.6 RECOMMENDATIONS

Completion of a pilot-scale demonstration of the bioreactor and landfill cover design concepts discussed in this report at an inactive, unlined DoD landfill is recommended. The demonstration would consist of retrofitting the landfill to operate in bioreactor mode, including collection and recirculation of leachate-contaminated groundwater, and installation of an ET cover (possibly with an organic layer). The test cell would be instrumented to measure infiltration rates below the ET landfill cover, and temporal changes in the chemical and geochemical characteristics of water in the vadose and saturated zones would be assessed. Based on an initial review of limited site data, landfills at three Air Force Bases are identified in Section 5 as potential pilot-test candidates.

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LIST OF ACRONYMS AND ABBREVIATIONS

ACAP	Alternative Cover Assessment Program
AFB	Air Force base
AFCEE/ERT	Air Force Center for Environmental Excellence/Technology Transfer Division
ARAR	applicable or relevant and appropriate requirement
bgs	below ground surface
BMP	best management practices
BOD	biological oxygen demand
BRA	baseline risk assessment
ºC	degrees Celsius
CA	chloroethane
CAA	Clean Air Act
CAH	chlorinated aliphatic hydrocarbons
CE	chloroethane
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CESQG	conditionally exempt small- quantity- generator
CF	chloroform
CFR	Code of Federal Regulations
CM	chloromethane
cm	centimeters
cm/sec	centimeters per second
CO_2	carbon dioxide
COC	contaminant of concern
COD	chemical oxygen demand
CSM	Colorado School of Mines
CT	carbon tetrachloride
CWA	Clean Water Act
2-D	two-dimensional
3-D	three-dimensional;
DCA	dichloroethane
DCE	dichloroethene
DNAPL	dense nonaqueous-phase liquid
DO	dissolved oxygen
DOC	dissolved organic carbon
DoD	Department of Defense
DoE	Department of Energy
ESTCP	Environmental Security Technology Certification Program
ET	evapotranspiration
Fe(III)	ferric iron
Fe^0	zero-valent iron
FR	*Federal Register*
FS	Feasibility Study
GWE	groundwater extraction

HELP	Hydrologic Evaluation of Landfill Performance
HRC™	Hydrogen-Release Compound
HW	hazardous waste
ITRC	Interstate Technology and Regulatory Cooperation Working Group
LF	linear foot
LTM	long-term monitoring
m^3	cubic meters
MC	methylene chloride
mg/L	milligrams per liter
Mitretek	Mitretek Systems
MNA	monitored natural attenuation
MSW	municipal solid waste
mV	millivolts
NA	natural attenuation
NAPL	nonaqueous-phase liquid
NAS	Naval Air Station
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
O&M	operations and maintenance
ORP	oxidation/reduction potential
Parsons	Parsons Engineering Science, Inc.
PCE	tetrachloroethene
POC	point of contact
PRB	permeable reactive barrier
RB/PB	risk-based/performance-based
RCRA	Resource Conservation Recovery Act
redox	reduction/oxidation
RI	remedial investigation
SACM	Superfund Accelerated Cleanup Model
SARA	Superfund Amendments and Reauthorization Act
SVE	soil vapor extraction
SWANA	Solid Waste Association of North America
TCA	trichloroethane
TCE	trichloroethene
TeCM	tetrachloromethane
TOC	total organic carbon
TS	treatability study
TVA	total volatile acids
UP	University of Pittsburgh
US	United States
USAF	US Air Force
USEPA	US Environmental Protection Agency
VC	vinyl chloride
WWTP	wastewater treatment plant
$year^{-1}$	per year
yd^3	cubic yards

SECTION 1

INTRODUCTION

1.1 PROJECT DESCRIPTION

Section 1.1 has been removed from this abbreviated version of the report. Please contact John Hicks (john.hicks@parsons.com) or Doug Downey (doug.downey@parsons.com) at Parsons in Denver, Colorado (303-831-8100) to obtain a copy of this section.

1.2 BACKGROUND

Hundreds of landfills on DoD installations have generated CAH plumes in groundwater. Surface covers, which are intended to provide a barrier to prevent direct contact with waste material and minimize or eliminate infiltration of precipitation through the waste material (i.e., leachate formation), are the current method of choice to manage human and ecological risks at these sites (Bagchi, 1990). The design elements intended to eliminate infiltration of water through the waste material are based on concerns about long-term leachate production and plume migration. Traditionally, these conventional landfill cover designs and management strategies have generally been imposed without consideration of how they may affect the biochemistry of CAH natural attenuation processes and the resulting risk reduction.

The conventional landfill closure strategy is expensive and not always warranted. A recent survey of USAF landfill-cover projects revealed an average cost of approximately $450,000 per acre for traditional Resource Conservation and Recovery Act (RCRA) covers (Hauser *et al*., 1999). In some cases, impermeable covers may actually impede natural attenuation processes by reducing the quantity of organic-rich leachates that promote reductive dechlorination or cometabolism of CAHs (Wiedemeier *et al*., 1999). In such cases, plumes will migrate farther and persist longer if natural attenuation processes are deprived of a continuing source of organic substrate. At some landfills, surface infiltration may accelerate the leaching of the "source" and reduce the time required for biological stabilization of the landfilled waste. In fact, recirculation of landfill leachate could be used to accelerate the source-leaching process and promote reductive dechlorination within a "closed loop" *in-situ* bioreactor (Pohland and Kim, 2000; Reinhart, 1998). This could reduce the time required for a CAH plume to stabilize and degrade, thereby reducing the long-term risk at the site. More rapid waste stabilization also will reduce long-term monitoring (LTM) costs because most regulatory agencies require rigorous post-closure monitoring activities as long as wastes pose a potential threat to water quality (US Environmental Protection Agency [USEPA], 1999a).

The 'enhanced-leaching' theory of landfill treatment challenges the conventional philosophy of 'infinite containment,' and could move the waste industry toward more permanent and efficient waste-management solutions. These solutions could include a greater reliance on evapotranspiration (ET) landfill covers using natural vegetation, and greater emphasis on natural attenuation rather than expensive leachate collection and treatment systems. Although enhanced-leaching methods are not appropriate for all landfills, they could be widely applied at many landfills where low levels of CAHs are the primary contaminants of concern (COCs).

The design approach proposed in this document maximizes the use of natural remediation and management techniques for landfill closures. By tailoring landfill closure requirements to site-specific natural-attenuation requirements, many CAH-contaminated landfills could select low-cost cover designs such as ET covers. ET covers can generally be installed and maintained for 50 percent of the cost of conventional RCRA covers. Hauser *et al*. (1999) estimated a potential savings of $500-$800 million if ET covers were substituted for RCRA covers at the nearly 600 USAF landfills located in the continental US. As an example of potential cost savings, F.E. Warren Air Force Base, Wyoming was recently was granted regulatory approval to install an ET cover that will result in cost savings of more than $12 million. The technical merits and performance of innovative landfill cover designs are being evaluated by numerous researchers including USEPA (Alternative Cover Assessment Program [ACAP]), Sandia National Laboratories, and the US Navy Facilities Engineering Service Center.

Additional cost savings are possible when landfill leachate is allowed to migrate and attenuate naturally or is recirculated in the landfill. Leachate collection and treatment systems can cost $200,000 to $400,000 per year to operate, and must be operated indefinitely at most landfills. At some landfills, leachate-recirculation techniques may be able to both contain and destroy CAHs, and reduce long-term operation and maintenance (O&M) costs for landfill-leachate treatment systems. For example, amortized costs estimated for a leachate-recirculation system at the Kootenai County Landfill in Idaho were 63 percent less than the cost estimate to treat the leachate off-site (Miller and Emge, 1997). Even if leachate collection and treatment are not required, LTM for conventional landfills may cost $50,000 to $100,000 annually. By allowing controlled infiltration and leaching to more rapidly stabilize the waste, LTM costs and liability also could be reduced.

Parsons has initiated dialogue with the Technology Transfer Division of the Air Force Center for Environmental Excellence (AFCEE/ERT) regarding the potential incorporation of the conceptual-design models presented in this document into the overall USAF and DoD landfill-design process. Parsons also has discussed this project with USEPA researchers, who are focusing on recirculation of leachate within lined RCRA landfills, ET and other low-cost cover designs, and developing better hydraulic models to predict the performance of various cover designs. Parsons' findings complement these ongoing initiatives, as well as the landfill management concepts discussed in a series of documents produced for the USAF by Mitretek Systems (Mitretek) (Weand *et al*., 1999; Boyer *et al.,* 1999; Gill *et al.,* 1999, and Hauser *et al*., 2001a), and it is expected that DoD installations with either closing or operating landfills will directly benefit from the results of this project. Based on the Mitretek studies, 86 percent of Air Force landfills have been closed for more than 20 years. Less than one percent have bottom liners, and

approximately 75 percent of the landfills do not have a final remedy in place. The Air Force has approximately 600 inactive landfills.

Parsons literature review focused on gathering landfill groundwater data to assess trends in contaminant distribution and geochemistry based on landfill age, geological setting, climate, etc. Our focus was on Air Force landfills that had previously been identified by the Mitretek studies. Parsons also interviewed several AFCEE project managers. Due to budget constraints, we were not able to duplicate this level of data gathering for the Navy or Army.

Parsons did contact representatives from the Army and Navy and discussed their landfill closure needs. Briefly, the Army has more landfills and much older landfills than the Air Force, many of which predate the use of chlorinated solvents. The exact number of Army landfills was unavailable. Explosives were generally disposed of in special landfills or burn pits, and are rarely encountered in domestic Army landfills (USACE, 2001). Army and Navy landfills established after 1940 frequently contain some chlorinated solvents and are very similar to Air Force landfills. The Navy has 429 inactive landfills in their environmental inventory, and approximately 25 percent of these landfills have CAH contamination (NAVFACHQ, 2002). Both of our Army and Navy contacts expressed interest in improving the landfill closure process (particularly for landfills with long-term requirements to collect and treat CAH-contaminated groundwater).

The results of this project will provide DoD environmental engineers with an additional perspective on landfill cover design and leachate treatment. This information will have universal application throughout the DoD, US Department of Energy (DoE), as well as at thousands of municipal landfills.

1.3 REGULATORY CONSIDERATIONS

Innovative landfill closure strategies can only be advanced in a regulatory environment that encourages alternative methods of risk control and reduction. According to USEPA (2001a), there are more than 3,000 municipal solid waste (MSW) landfills currently operating in the US. MSW landfills typically receive non-hazardous household, commercial, institutional and industrial waste; hazardous household waste; and conditionally exempt small-quantity-generator (CESQG) hazardous waste (USEPA, 2000c). The design and operation of most of these MSW landfills comply with current solid waste management regulations, which set forth minimum performance and design standards to ensure that "no reasonable probability of adverse effects on health or the environment" will result from solid waste disposal facilities (Title 40 of the Code of Federal Regulations [CFR], Part 258). However, there is growing recognition that these current minimum performance and design standards merely extend the time required for the "managed" solid waste to adversely impact underlying soil and groundwater quality (Cherry, 1990; Lee and Jones-Lee, 1997). In fact, the nature and magnitude of the future releases from the landfill are directly related to the amount of organic waste not yet decomposed.

Additionally, the number of "closed" (or abandoned) MSW landfills, many of which are unlined, poorly capped, and located on alluvial materials, could be as high as 100,000

(Suflita *et al.*, 1992). Chemical constituents within the solid wastes and substances produced by the breakdown of these wastes can be leached from these unlined landfills by infiltrating water. The leachate is ultimately transported to the water table, thus adversely affecting groundwater quality, and potentially exposing human and/or ecological receptors to COCs. Clearly, understanding the complex hydrogeological and biochemical processes governing the fate of these landfill-derived contaminants is critical to developing management strategies that truly are protective of human health and the environment for both operational and "closed" MSW landfills.

DoD is responsible for the management of thousands of non-operational MSW landfills. The USAF alone is currently responsible for about 600 landfills, of which approximately 375 have not yet been remediated. Over the years, military bases used landfills to dispose of solid wastes, including municipal waste, construction debris and rubble, industrial waste, cleaning solvents, paint strippers, and pesticides (Gill *et al.*, 1999). Less than 1 percent of these landfills have liners below the waste (Hauser and Weand, 1998). More than 80 percent of these landfills have not been operational since the late 1980s, when the use of contract waste-disposal services became more widespread (Gill *et al*., 1999).

The objective of this section of the report is to review the current regulatory requirements governing the management of MSW landfills, particularly during the closure and post-closure periods. It is important to understand the existing regulatory paradigms for MSW landfills when developing alternative MSW landfill management strategies for implementation, because these requirements 1) clearly define the current technical areas of concern that must be adequately addressed by any alternative approaches, and 2) ultimately affect how readily alternative strategies may be accepted by federal and state regulators and the public. Additionally, this section reviews current regulatory trends that may in fact support the use of alternative MSW landfill management strategies.

1.3.1 NCP and CERCLA

The Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) of 1980, also known as Superfund, directed USEPA to identify abandoned or uncontrolled hazardous-waste sites, and to cleanup the worst of those sites. The Agency carries out these responsibilities through the Superfund response process, as set forth in the National Oil and Hazardous Substances Pollution Contingency Plan (NCP). Section 300.430(a)(iii)(B) of the NCP establishes the expectation that engineering controls, such as containment, will be used for waste that poses a relatively low, long-term threat or where treatment is impracticable. In fact, the preamble to the NCP identifies MSW landfills as a type of site where treatment of the waste may be impracticable because of the size and heterogeneity of the contents (55 *Federal Register* [FR] 8704). However, the NCP also establishes the expectation that treatment should be considered for MSW landfills for identifiable areas of highly toxic and/or mobile material that could pose a potential principal threat. This provision could be an important argument in terms of pursuing treatment-based management strategies for MSW landfills.

USEPA's authority to initiate a Superfund response is triggered by the release or threatened release of a hazardous substance into the environment. To create the list of

regulated hazardous substances, the US Congress incorporated many substances regulated under other federal environmental laws, such as the Clean Air Act (CAA), the Clean Water Act (CWA), and RCRA. Within the Superfund Amendments and Reauthorization Act (SARA) of 1986, Congress essentially enacted as law USEPA's policy to use other environmental laws to guide response actions. SARA added CERCLA Section 121(d), which stipulates that the remedial standard or level of control for each hazardous substance, pollutant, or contaminant be at least that of any applicable or relevant and appropriate requirement (ARAR) under federal or state environmental law. A requirement that is not legally applicable may be relevant and appropriate if it addresses problems or pertains to circumstances similar to those encountered at a Superfund site. Compliance with relevant and appropriate requirements is determined at the discretion of the involved regulators. Consequently, the identification of which environmental laws to use to establish performance requirements will—in concert with the results of the baseline risk assessment (BRA)—define the types of response actions that could reasonably be undertaken at a specific CERCLA site.

1.3.1.1 ARARs and MSW Landfill Management

RCRA is the controlling federal law for both MSW and hazardous waste (HW) landfills. Additionally, 49 of the 50 states have been granted the authority by the USEPA to regulate their own solid-waste programs, because they have adopted requirements that are at least as stringent as those set forth in the federal regulations. Although the remediation of old landfills is not addressed specifically under RCRA, these sites are typically regulated under the authority of CERCLA with RCRA often becoming the source of potential activity-specific ARARs (i.e., closure and post-closure care requirements). Federal and/or state RCRA landfill criteria (see below) are almost routinely identified as action-specific ARARs for CERCLA MSW landfills, even though RCRA itself provides considerable latitude in determining applicability at landfill sites that ceased operations prior to October 9, 1991. USEPA (1991) even specifically noted that RCRA landfill closure requirements should be evaluated as potentially relevant and applicable requirements only for CERCLA-regulated MSW landfill sites.

The designation of a landfill as a MSW or HW landfill also will greatly influence the regulatory and technical requirements that could be identified as ARARs for a particular site. In general, the federal and state requirements governing HW are much more stringent than those applied to MSW. Although no presumptive remedy for HW landfills has been established (see below), the nature and intent of the HW landfill regulations may pose a significant impediment to alternative landfill-management strategies.

A broad framework for the CERCLA remedial investigation/feasibility study (RI/FS) and remedy-selection process has been created through the NCP and the USEPA RI/FS guidance (USEPA, 1988 and 1991). This framework provides for the waiver of ARARs in certain situations. For example, an ARAR may be waived if compliance with the requirement would result in greater risk to human health and the environment than non-compliance (e.g., if implementation of an ARAR effectively postpones potential adverse exposure and/or environmental effects). Additionally, an ARAR may be waived if an alternative design or method of operation can produce equivalent or superior results, in terms of the degree of protection afforded, the level of performance achieved, long-term protectiveness, and the time required to achieve beneficial results. Consequently, any

effort to pursue implementation of alternative response actions at MSW landfills should include clear documentation on at least these issues.

1.3.1.2 Presumptive Remedy for CERCLA MSW Landfills

In 1992, USEPA introduced the Superfund Accelerated Cleanup Model (SACM) to accelerate all phases of the CERCLA remedial process. The presumptive-remedy initiative was one tool included in the SACM. Presumptive remedies are preferred technologies for common categories of sites based on historical patterns of remedy selection and USEPA's scientific and engineering evaluation of performance data on technology implementation. Presumptive remedies are expected to be used at all appropriate sites. USEPA (1993b and 1995) estimated that the presumptive remedy for MSW landfills would be an appropriate response action for at least 75 percent of the landfills regulated under CERCLA. The USEPA noted that this presumptive remedy also should be applied to appropriate MSW landfills subject to the RCRA corrective action program (USEPA, 1995 and 1996). The MSW presumptive remedy is not applicable for HW landfills.

USEPA (1993a) established source containment as the presumptive remedy for MSW landfill sites regulated under CERCLA in September 1993. The containment presumptive remedy includes the following components, as appropriate on a site-specific basis:

- Landfill cap,
- Source-area groundwater control to contain plume,
- Leachate collection and treatment,
- Landfill gas collection and treatment, and
- Institutional controls to supplement engineering controls.

The presumptive containment remedy does not include long-term groundwater response actions, which may be identified as supplemental to the presumptive remedy or addressed as a separate action (USEPA, 1993b).

In a 1996 directive, USEPA (1996) clearly stated that this presumptive remedy should also be applied to all appropriate military landfills. The presumptive remedy for MSW landfills is based on language in the NCP, wherein USEPA concluded that 1) these sites often pose a low-level threat rather than a principal threat because they contain municipal and, to a lesser extent, hazardous wastes; and 2) the volume and heterogeneity of waste often makes treatment impractical. Additionally, USEPA (1993b and 1995) notes that, given the knowledge base available in 1993, the containment presumptive remedy provides the means to attain the following technical goals:

- **Minimize infiltration**: the objective is to prevent water from percolating through the waste and forming leachate, which can pollute underlying soil and groundwater.

- **Isolate wastes**: the objective is to minimize the potential for direct receptor contact with exposed waste at the ground surface.

- **Control landfill gases**: the objective is to provide for the controlled release and/or capture of explosive or toxic gases that can create a potential hazard in the vicinity of a landfill.

The presumptive-remedy approach was designed to streamline the RI/FS process, which can result initially in significant time and cost savings (USEPA, 1997a). However, it is important to note that the long-term costs associated with inappropriately applying the presumptive remedy may far outweigh any initial cost savings realized by minimizing site characterization, streamlining the risk assessment, and not conducting a thorough evaluation of alternative approaches. Presumptive remedies also provided the USEPA (2000f) with a mechanism to promote consistency in remedy selection and implementation.

1.3.2 RCRA's 1991 Solid-Waste-Disposal Criteria

RCRA was enacted to regulate ongoing waste-management activities in order to encourage resource conservation and to protect human health and the environment. As specified in RCRA, the federal role is to establish overall regulatory direction through the provision of minimum nationwide standards for MSW landfills. On October 9, 1991, the USEPA (1991) issued *Criteria for Municipal Solid Waste Landfills* (40 CFR 258; 56 FR 50978). These criteria established minimum national performance standards necessary to ensure that "no reasonable probability of adverse effects on health or the environment" will result from solid-waste disposal facilities. In terms of waste-management activities using a landfill unit, RCRA sets forth both performance and design specifications for construction, operation, and closure, including surface-water requirements, a groundwater-contamination detection/monitoring program, a closure-system assessment/ monitoring program, closure criteria, and post-closure care requirements.

RCRA regulations strictly apply to MSW landfills that received waste on or after October 9, 1993. Only the final cover requirements apply to those MSW landfills that received waste between October 9, 1991 and October 9, 1993. Landfills that stopped accepting waste before October 9, 1991 are not required to comply with the RCRA criteria. This is the reason why the RCRA MSW regulations are not to be identified or *de facto* treated as legally applicable requirements during CERCLA response actions. Any contamination of groundwater beneath and downgradient from a regulated MSW landfill may trigger corrective-action requirements under RCRA, which are addressed under a separate USEPA program.

As described previously, RCRA enforcement authority is delegated to the states as each state adopts equal or more stringent regulations than those contained in federal rules and regulations. Generally, most state regulations closely follow RCRA regulations.

1.3.2.1 Performance and Design Specifications for Liner Systems

The containment approach to MSW landfill "closure" focuses on minimizing infiltration of precipitation so that water cannot percolate through the waste and carry soluble wastes downward into groundwater. This approach hinges on the assumption that

wastes could be indefinitely isolated from the environment using engineered controls (e.g., liner systems, barrier caps, leachate collection systems). Consequently, all modern landfills are constructed with the end goal of "closure" by containment in mind.

Pursuant to the 1991 MSW landfill criteria, all regulated landfills must have liner systems beneath the waste to prevent the downward movement of leachate and contamination into the environment. The criteria set forth two methods for complying with bottom-liner requirements for landfills. The first is a performance standard, while the second is a specific design standard. Under the performance standard set forth in 40 CFR 258.40(a)(1), the liner may be constructed so that the concentration values for specified monitored constituents will not be exceeded in the uppermost aquifer at the relevant point of compliance (POC). The POCs for modern landfills regulated under the RCRA 1991 criteria are usually no more than 150 meters (490 feet) from the waste-management-unit boundary. The second method for compliance is to install a liner system that meets the specific design criteria described in 40 CFR 258.40(a)(2) and set forth in 40 CFR 258.40(b). The federal minimum liner system includes a single composite liner; many states require at least a double composite liner.

Most military landfills were constructed without any form of bottom-liner system (Weand *et al*., 1999). This deficiency will have profound ramifications on the management strategies that may be appropriate for these sites (and arguably for all landfill sites, since no liner system is expected to provide indefinite performance).

1.3.2.2 Minimum Final Cover Requirements

At the time RCRA was implemented, barrier-type covers using multiple low-permeability layers were considered the most permanent and protective landfill cover options. While current regulations allow for some design flexibility, both MSW and HW final cover specifications include specific permeability requirements reflecting this prejudice. Final covers for HW landfills must be constructed to minimize the migration of liquids through the "closed" landfill by providing a permeability less than or equal to the bottom liner or natural subsoils [see 40 CFR 264.310(a)(5)]. The closure requirements for HW landfills include a leachate-collection system and a leak-detection system.

Similarly, for MSW landfills, the general goal of a "closure" cover is to minimize infiltration. Consequently, the RCRA criteria specify that the final cover for MSW landfills provide a permeability less than or equal to the permeability of any bottom liner system or the natural subsoils, or in any case have a permeability no greater than $1x10^{-5}$ centimeters per second (cm/sec) [see 40 CFR 258.60(a)(1)]. Generally, the final cover should be constructed of an infiltration layer composed of a minimum of 18 inches of earthen material (to minimize the flow of water into the "closed" landfill), and an erosion-control layer composed of a minimum of 6 inches of earthen material capable of sustaining plant growth (to slow the disintegration of the cover). Unlike the HW landfills, however, the criteria provide greater flexibility in terms of selecting an alternative final cover design that includes an infiltration layer that achieves an equivalent reduction in infiltration by methods other than permeability [see 40 CFR 258.60(b)(1)]. Clearly, some methods of facilitating *in situ* degradation of wastes (within the landfill

source area, which is the subject of all MSW regulations) would be at odds with the implicit intent of waste isolation and infiltration minimization.

Post-closure requirements for MSW landfills include continued operation of the leachate collection and removal system until leachate is no longer detected, and maintenance and sampling of the groundwater monitoring system for at least 30 years. In the past few years, after implementing post-closure requirements at many sites, USEPA and its state counterparts have recognized the need to develop a rational and technically based method for terminating post-closure care requirements at landfill facilities (USEPA, 2000d).

1.3.2.3 Provisions Governing Leachate Recirculation

There is a growing body of evidence that suggests that the containment presumptive remedy prescribed by current regulations merely postpones the threat that MSW landfills pose to human health and the environment (Barlaz *et al.*, 2000; Cherry, 1990; Lee and Jones-Lee, 1997; Pacey *et al.*, 1999). Eventually, the liner system and/or low-permeability "closure" cover will fail, allowing moisture to enter the "entombed" wastes. Basically, the MSW landfill management approach currently being implemented in the US is fundamentally flawed, because the "closed" landfill may pose a threat to underlying soil and groundwater for as long as the wastes in the landfill will be a threat.

An alternative approach for closing both modern and unlined MSW landfills is to abandon the effort to keep the wastes dry, and to begin more aggressive treatment/management of the waste materials and polluted groundwater resulting from leachate generated in the landfill. Under this approach, the landfilled waste would be deliberately treated to enhance moisture, to promote both fermentation and leaching. The additional leachate produced during this waste-treatment period could be managed using appropriate remediation techniques, such as natural attenuation or engineered controls. This approach may be allowed in situations where groundwater contamination is already present (i.e., at unlined MSW landfills), provided that the treatment approach is controlled and targets the source, the plume size remains stable, risks to downgradient receptors are controlled, and adequate contingency measures are in place (USEPA, 1999b and 2001d). A more thorough review of these possibilities is a primary topic of this report.

Such an alternative MSW landfill management strategy would require profound changes in the current regulatory paradigm in order to be implemented. In addition to the basic philosophy change from containment to treatment, the MSW criteria have generally been interpreted to prohibit the addition of bulk liquids to the contained waste [see 40 CFR 258.28 and 258.40(a)(2)]. However, if the liquid addition is presented as a "treatment amendment," it is possible that such an approach may be permissible under certain conditions (i.e., within MSW landfills with specific bottom-liners). Similarly, leachate recirculation, which may be an important component of efforts of enhance anaerobic fermentation of CAHs, is allowed only at MSW landfills with a composite liner and leachate-recirculation system, as described in 40 CFR 258.28(a)(2).

The recirculation of leachate currently is not allowed in landfills that have an alternative liner design, even if the design meets the performance standard in 40 CFR

258.40(a)(1), although the USEPA (2000b and 2000c) is currently revisiting this issue. USEPA (1997b) has focused considerable effort on defining best management practices (BMPs) for various waste-treatment technologies that minimize cross-media impacts. Deliberate leaching from source-area wastes into underlying soils and groundwater, even with the intent to facilitate treatment, may not be considered a BMP. However, recent initiatives by USEPA to pursue risk-based/performance-based (RB/PB) cleanups may provide an excellent forum to explore these alternative MSW landfill management strategies.

1.3.3 Regulatory Trends in Support of Alternative MSW Closure Strategies

As evidenced by the previous discussion, many of the current regulations and standard implementation approaches to long-term MSW landfill management ("closure") may prove to be a significant impediment to innovative management strategies that do not build upon the traditional containment approach. However, recent actions by USEPA indicate that the regulatory climate may be changing, thus making it more likely that regulators would and could be able to approve more innovative MSW landfill closure strategies. Following is a brief synopsis of evolving USEPA program policies, interpretations, and research interests that may have a direct impact on the use of innovative MSW landfill management strategies that include natural attenuation. This overview is intended to highlight federal program changes only. As noted previously, RCRA authority to regulate MSW landfills has been delegated to most individual states; consequently, specific state regulations and policies must be consulted to determine the likelihood that any alternative management strategy would be favorably received for implementation review at any particular site.

1.3.3.1 RCRA Reforms: Risk-Based Corrective Action Paradigm

To more effectively meet the challenging goals of and accelerate the pace of cleanups completed under the authority of the RCRA corrective action program, USEPA introduced the RCRA Cleanup Reforms in 1999, and is implementing additional Reforms in 2001 (USEPA, 2001b). The 1999 and 2001 Reforms build upon actions taken by USEPA and the states in recent years to expedite remediation efforts at sites under the RCRA corrective action program (e.g., groundwater contamination originating from "old" unlined MSW landfills). The RCRA Cleanup Reforms of 2001 promote use of innovative approaches, accelerate changes in regulatory culture, foster better connections with communities affected by cleanups, and focus on remediating to capitalize on long-term development potential. Clearly, all of these areas may become key drivers in successful implementation of any innovative MSW landfill management strategies.

USEPA has been focusing on RB/PB corrective-action approaches since the *1996 Advanced Notice of Proposed Rulemaking (ANPR), Corrective Action for Solid Waste Management Units at Hazardous Waste Management Facilities* (61 FR 19432). RB/PB corrective action encourages technical and administrative innovation to achieve environmentally protective cleanups on a facility-specific basis. Specifically, the RB/PB corrective-action approach directs both regulators and the regulated community to focus on end goals (e.g., sustainable and demonstrated protection of human health and the environment), rather than on some process aimed at achieving those goals. This process thus may be used to challenge the need to follow the classical ARAR approach to

determine MSW landfill closure requirements, because it encourages the regulator/owner to select the most technically sound and cost-effective alternative to address the risks at a particular landfill (Gill *et al*., 1999; USEPA, 2001b). Consequently, the RB/PB corrective action should hinge on identifying site-specific performance standards to prescribe the scientific, technical, and administrative requirements that must be fulfilled in order to complete the corrective action (USEPA, 2000f).

This shift in regulatory priorities appears to create a favorable framework within which to explore innovative management strategies for MSW landfills that are intended to degrade source-area wastes and/or minimize the potential long-term threat of landfilled wastes. A RB/PB landfill evaluation completed under the evolving RCRA corrective action program could then be technically based on the specific conditions at a landfill. This evaluation would need to be tailored to a specific site, and could not be based on the "streamlined" risk assessments that conventionally accompany a presumptive-remedy determination (USEPA, 1993b). Once these technical performance requirements are determined and accepted by the public and regulatory community, any particular landfill remediation scenario that meets them - including alternative management approaches – could be selected and implemented. Consequently, this program may provide yet another avenue to challenge the presumptive remedy for MSW landfills set forth in current guidance. Note that the presumptive-remedy guidance already clearly provides for the selection of an alternative closure approach, if the site is deemed "not appropriate" for the general containment strategy; however, USEPA (1991, 1993b, and 1997a) had previously estimated that the containment presumptive remedy would be applicable for more than 75 percent of all known MSW landfills.

1.3.3.2 USEPA's Position on Monitored Natural Attenuation

USEPA (1999b) also has recently reiterated the viability of monitored natural attenuation (MNA) as an alternative means of achieving remediation objectives that may be appropriate for specific, well-documented site conditions. The potential impacts of natural attenuating processes on contaminant release and migration were originally discussed in the NCP, and are to be addressed in the CERCLA RI/FS process (e.g., USEPA, 1991). USEPA defines MNA as the reliance on natural attenuation processes (within the context of a carefully controlled and monitored site-cleanup approach) to achieve site-specific remediation objectives within a timeframe that is reasonable compared to that offered by other more active methods. The natural attenuation processes include a variety of physical, chemical, or biological processes that, under favorable conditions, act (without human intervention) to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in soil or groundwater. These *in situ* processes can include biodegradation, dispersion, dilution, sorption, volatilization, decay, and chemical or biological stabilization, transformation, or destruction of contaminants. Section 2 of this report includes a brief overview of the general microbially mediated natural attenuation processes that may be operating in groundwater beneath and downgradient from a MSW landfill from which a release of chlorinated compounds has occurred.

Per a recent policy statement, USEPA (1999b) expects that sole reliance on MNA as the remediation approach will be appropriate only for sites that have a low potential for contaminant migration. However, as the scientific understanding of natural attenuation

processes continues to evolve, USEPA recognizes that it may now be possible — on a site-specific basis — to demonstrate that these processes could be sufficient to achieve significant remediation objectives (e.g., preventing further migration of contaminants from source areas, preventing further migration of groundwater contaminant plumes, and/or reducing contamination in soil or groundwater to specified cleanup levels appropriate for current or potential future uses). USEPA anticipates that some form of source control and long-term performance monitoring will be fundamental components of any MNA remedy. It is important to note that USEPA (1999b), in its MNA policy, encourages the consideration of innovative technologies for source control or "active" components of the remedy, which may offer greater confidence and reduced remediation timeframes at modest additional cost. This statement could be used to support the use of covers that promote "enhanced leaching" of MSW landfill source materials as an innovative source-control method.

One of the primary concerns that USEPA (1999b) has expressed with regard to MNA remedies is that, at some sites, the same biochemical conditions and processes that lead to biodegradation of CAHs and petroleum hydrocarbons can chemically transform naturally occurring minerals in the aquifer matrix into forms that are more mobile and/or more toxic than the original materials. Additionally, some natural attenuation processes may result in the creation of transformation products that are more toxic and/or mobile than the parent contaminant (e.g., degradation of TCE to VC). Clearly, the potential for creation of toxic transformation and/or secondary byproducts should be evaluated to determine if continued reliance on these processes would be appropriate and protective over the long term.

Bioremediation-based technologies are being used with increasing frequency to remediate contaminated media at hazardous waste sites because, compared with other remediation technologies, they are often less expensive and more acceptable to the public. A recent review of 104 Superfund remedial action projects indicated that 20 of 80 *in situ* projects for groundwater treatment relied on either natural and/or augmented bioremediation processes (USEPA, 2001e). This review indicated that groundwater at several landfill and/or disposal areas was currently in the process of being bioremediated through the use of recirculation-cell operations (essentially pump-and-treat with upgradient reinjection to promote *in situ* biodegradation). The next subsection briefly summarizes the USEPA's position on the use of groundwater capture and reinjection as a treatment method.

1.3.3.3 USEPA's Position on Reinjection as a Treatment Approach

USEPA (2000g) recently clarified that reinjection of treated groundwater to promote *in situ* treatment of groundwater contamination is allowed under RCRA section 3020(b), provided that: 1) groundwater is treated prior to reinjection, 2) the treatment is intended to reduce contaminant concentrations (either before or after reinjection), 3) the approach is protective of human health and the environment, and 4) the injection is part of a CERCLA or RCRA response action intended to remediate the groundwater contamination. This is an important clarification because it considers amending extracted groundwater (with nutrients, oxygen, etc.) as treatment, and it specifically recognizes that contaminant reductions can be achieved after reinjection (via biodegradation). Using a broad interpretation, this clarification memorandum may make it possible for USEPA to

consider the recirculation of collected groundwater through an unlined MSW landfill, which alone may qualify as "treatment" (with or without engineered amendments), as an allowed source/groundwater-remediation method as long as the approach is protective of human health and the environment. The USEPA intended this clarification to apply to groundwater bioremediation efforts, which would be in keeping with alternative MSW landfill-management strategies that promote natural attenuation mechanisms as part of long-term stabilization and/or remediation activities.

1.3.3.4 Project XL: Leachate Recirculation, Alternative Liner Designs, and Bioreactors

Emerging bioreactor landfill technology processes can be an environmentally favorable step forward in the final disposal of organic waste materials in MSW landfills. The bioreactor landfill differs from the design specifications set forth in the 1991 federal MSW landfill criteria because it receives managed liquid additions to augment waste stabilization. The bioreactor landfill is generally defined as a landfill operated to transform and more quickly stabilize the readily and moderately decomposable organic constituents of the waste stream by purposeful control to enhance microbiological processes (Pacey *et al*., 1999). Bioreactor landfills often employ controlled liquid addition with leachate recirculation within lined waste-management cells, alternative cover designs, and state of the art landfill gas collection systems (Solid Waste Association of North America [SWANA], 2000; USEPA, 2000c).

As discussed previously, current MSW landfill regulations generally encourage landfills to remain dry. Although the federal regulations do not expressly bar the use of amendments, many states have interpreted 40 CFR 258.28 to mean that any form of liquid addition (other than leachate and condensate collection and recirculation, as expressly provided) is not allowed. However, the federal MSW landfill regulations may allow the addition of water and other amendments if they are part of waste-stabilization efforts, provided minimum specified bottom-liner requirements are met.

Recent activities by USEPA indicate that the regulatory climate related to bioreactors may be evolving. USEPA has long supported efforts to develop innovative solid waste management strategies. For example, they are active technical participants in Project XL, which was initiated by President Clinton to explore innovative solutions to pressing environmental problems. Under this effort, several sites have been selected and funded to explore the use of bioreactor-type technologies at large-scale, state-of-the-art MSW landfills (USEPA, 2000a, 2001c, and 2001d). Some of these recirculation projects will directly address the CAH degradation potential in lined landfills, which is the first step in expanding this research program to unlined landfills (Kramer, 2001). Additionally, in the past 2 years (and possibly as a result of these upcoming evaluations), USEPA (2000c) appears to be seriously considering significant revisions to the *Criteria for Municipal Solid Waste Landfills* (40 CFR Part 258) with respect to the operation of landfills as more advanced bioreactors. Although these activities may have limited immediate applicability to unlined MSW landfills, they may serve as a springboard for technical discussions on how the bioreactor concepts may (or may not) apply to other situations.

1.3.3.5 Proposed Revisions to Criteria for MSW Landfill Closure Regulations

USEPA (2000d) is in the process of revising the federal MSW landfill regulations because new issues and technologies have emerged since the guidance was last revised in 1991. Recommended changes to the closure guidance should be available for review in December 2001. Based on a recent summary of the scope of the considered revisions, it appears that the revisions will be limited to evolving cover and liner technologies. Containment with performance monitoring appears to be the primary MSW landfill long-term management approach to be addressed in federal regulations.

SECTION 2

NATURAL ATTENUATION OF CHLORINATED ALIPHATIC HYDROCARBONS IN LANDFILLS

Section 2 has been removed from this abbreviated version of the report. Please contact John Hicks (john.hicks@parsons.com) or Doug Downey (doug.downey@parsons.com) at Parsons in Denver, Colorado (303-831-8100) to obtain a copy of this section.

SECTION 3

SUMMARY OF LITERATURE REVIEW

This section summarizes recent technical and scientific information that may be relevant to establishing an alternative, state-of-the-art MSW landfill management strategy that provides for treatment of source areas and/or groundwater affected by chemical releases. Discussions in this section are organized into four general research areas: 1) landfill chemistry studies, 2) case studies on the effectiveness of natural attenuation processes on chlorinated solvents within and/or released from landfills, 3) alternative landfill management concepts, and 4) alternative landfill cover designs. This literature review directly supports the evaluation of recommended alternative MSW landfill design and management strategies, which are presented in Section 4 .

3.1 LANDFILL CHEMISTRY STUDIES

A critical element in understanding how landfilled materials may be treated *in situ* is determining which bio/geochemical processes naturally dominate the environment within and adjacent to the landfill. This information is directly relevant to the transport and fate of organic contaminants within the source materials and underlying groundwater. A number of studies, which rely on *in situ* monitoring methods and/or chemical analysis of collected leachate samples, have been completed for unlined MSW landfills to evaluate the bio/geochemical conditions within the landfill (e.g., Reinhart and Pohland, 1991; Lyngkilde and Christensen, 1992a and 1992b; Chu *et al.,* 1994; Mirecki and Parks, 1994; Lee *et al.,* 1995; Nielsen, 1995; Brasaemle *et al.,* 1997; Basberg *et al.,* 1998; Ludvigsen *et al.,* 1998; Onay and Pohland, 1998; Albrechtsen *et al.,* 1999; Johnson *et al.,* 1999; Trabelsi *et al.,* 2000).

3.1.1 Measured or Implied Redox Conditions

In general, these studies suggest that redox conditions vary spatially within MSW landfills, with the most reducing conditions located immediately adjacent to the source of organic matter undergoing *in situ* degradation. As the distance from the source increases, the redox conditions evolve from strongly reducing (methanogenic) to oxidizing (oxic). This spatial distribution of redox conditions can be limited to the boundaries of the landfill itself (if no releases have occurred), or it can define the redox pattern of the entire area impacted by the landfill (i.e., the landfill source area as well as any groundwater impacted by releases).

The geochemical conditions within a landfill can be assessed by measuring indicator compounds for various redox reactions. For example, aerobic conditions have been inferred by measuring dissolved oxygen (DO) concentrations within landfill leachate

(e.g., Lee *et al.*, 1995). Similarly, the presence of sulfate-, iron-, and nitrate-reducing conditions, and methanogenic conditions has been inferred from the measurement of indicator chemical species (e.g., sulfate, ferrous iron, nitrate/nitrite, and methane). These types of evaluations suggest that the geochemical conditions within and downgradient from an unlined MSW landfill follow a typical redox zone sequence that ranges from methanogenic close to the landfill source to sulfate-reducing, iron-reducing, manganese-reducing, and nitrate-reducing to aerobic at the outskirts of the plume (Lyngkilde and Christensen, 1992a and 1992b; Christensen *et al*., 1994; Nielsen, 1995; Ludvigsen *et al.*, 1998; Albrechtsen *et al.*, 1999). The presence of methanogenic conditions near the landfill source coincides with potential sources of methane gas generation, which may require some form of gas collection to prevent safety hazards and other adverse conditions under certain circumstances.

The sequence of redox zones is generally limited by the presence of oxidized species within the landfill material and/or aquifer sediment. Specifically, if some of the electron acceptors involved in the redox reactions are missing (or depleted with no potential for replenishment, as may occur under engineered "dry" landfill conditions), the corresponding redox zones will be missing. These sequential redox zones may exhibit a substantial capacity to degrade specific organic compounds such as CAHs.

3.1.2 Chemical Characteristics of Landfill Leachate

In addition to direct redox measurements, chemical analysis of produced/collected leachate and leachate-impacted groundwater can be an important source of information about the chemical conditions predominating within and downgradient from a MSW landfill. The characteristics of landfill leachate are highly variable depending on the composition of the solid waste, rate of water infiltration, refuse moisture content, and landfill design, operation, and age (Reinhart and Pohland, 1991). Initially, leachate may contain extremely high concentrations of organic (Albaiges *et al.*, 1986; Schultz and Kjeldsen, 1986) and inorganic constituents. Over time, however, the concentrations of leachate constituents typically decline as a result of leaching (washout) and/or biological activity (Onay and Pohland, 1998; Trabelsi *et al.*, 2000).

An investigation of leachates obtained from landfills operated from 1 to 20 years indicated that the relative abundance of high-molecular-mass humic-like substances decreased with age, while intermediate-sized fulvic materials showed significantly smaller decreases (Chian, 1977). Chian and DeWalle (1976) reported that the pH of leachate increased with time due to decreasing concentrations of partially ionized free volatile fatty acids. Chian (1977) and Harmsen (1983) demonstrated that organic matter in leachate generated from older landfills mainly consisted of high-molecular-weight hydroxyaromatic substances (e.g., humic acid, fulvic acid, tannic acid, gallic acid), which are relatively inert to biological degradation.

Although the change in organic substances over time may be anticipated, changes in the concentration of inorganic constituents that may not be directly involved in redox reactions also have been reported (Basberg *et al.*, 1998). Elevated concentrations of various metals have been reported in groundwater influenced by leachate from an unlined MSW landfill (Basberg *et al.*, 1998). These metals may have been mobilized from the aquifer sediments due to the radically altered redox conditions resulting from degradation

of organic compounds. The change in basic inorganic chemistry can be a useful initial indicator of the occurrence and scale of organic releases from MSW landfills (Barr and Fernandez, 1980). Additionally, it has been commonly observed that the concentrations of ammonia and nitrogen increase rapidly within a landfill over time (e.g., Hartmann and Hoffmann, 1990; Chu *et al.,* 1994).

Pilot-scale data show that preferential flow paths in landfill material strongly affect the leachate chemistry and transport. More rapid flow occurs through large pore spaces and slower flow through smaller pores (Oman and Rosquist, 1999). The preferential flow paths create a complex set of reactions including solubility, sorption, and complexation that control the inorganic composition of the leachate (Johnson *et al.,* 1999).

3.2 NATURAL ATTENUATION OF CHLORINATED SOLVENTS IN LANDFILL LEACHATE (CASE STUDIES)

Reliance on natural attenuation processes to provide *in situ* treatment of landfilled waste and/or remediation of leachate plumes in groundwater has not yet been seriously considered as part of alternative MSW landfill management strategies. Data collected at various MSW landfills worldwide (particularly in Denmark) suggest that natural attenuation processes may serve to complement other management approaches. However, the site characterization and performance monitoring requirements for MSW landfills using natural attenuation as part of a holistic management approach would likely be more complicated and demanding than those typically required for non-landfill waste sources (Christensen *et al.,* 2000). This increased level of characterization and monitoring can be expected for MSW landfills, because the unique and special features of these sites may prompt site-specific modification of the approaches and protocols commonly applied at non-landfill fuel and solvent plumes (USEPA, 1998b; Wiedemeier *et al.,* 1999).

The total mass of CAHs that may be released in MSW landfill leachate depends on the source mass and distribution within the landfill, the adequacy of containment, and the environmental setting of the landfill. For instance, much of the original source mass may have been depleted by prior dispersion and/or biotransformation in older unlined landfills because there was sufficient moisture to support biological activity and leachate formation. In comparison, in newer or more actively stabilizing landfills, significant CAH mass may remain. Hence, the long-term management/remediation approach developed for either of these cases (or across the spectrum of potential scenarios bounded by these cases) would likely be different, but capable of being based on current understanding of biotransformation principles.

3.2.1 Landfill Plume Characteristics

Available data indicate that most groundwater leachate plumes originating from MSW landfills are relatively short and narrow (Christensen *et al.,* 1994 and 2000). The leachate plumes are commonly dominated by low ORPs, indicating that anaerobic natural attenuation processes may be operant. Such reducing conditions support several microbially mediated redox reactions, including methanogenesis, sulfate reduction, iron reduction, manganese reduction, and denitrification. As noted previously, the low redox conditions prevailing beneath MSW landfills also may result in the dissolution of metals

(e.g., iron) from aquifer sediment. Once the leachate plume is beyond the immediate environs impacted by the landfill source, the groundwater quickly returns to its natural oxidizing state.

Laboratory and field-scale experiments have shown that several organic compounds may slowly biodegrade in leachate-contaminated groundwater (Christensen *et al*., 2000). However, nearly all of the leachate plumes comprehensively studied to date have been in naturally aerobic, sandy aquifers. The potential for effective biodegradation of organic compounds such as CAHs in groundwater characterized by anaerobic conditions, and/or limited constituents that may participate in known paired redox reactions (e.g., iron, manganese, nitrate), has not been investigated in any great detail.

Although landfills tend to persist under anaerobic conditions during the majority of their active stabilization period, the generation of CAHs (or lack thereof) can be assessed by examination of samples of the gas and leachate. Gas-phase CAHs have been detected in measurable concentrations in close proximity to the landfill surface (Deipser and Stegmann, 1994), and such evidence is useful in helping to determine the biochemical state of the landfill.

There are several features of MSW landfills that may complicate the characterization and/or LTM of natural attenuation processes (Christensen *et al*., 2000). The sources of landfill leachate plumes are generally complex, difficult to characterize, and not amenable to targeted removal activities. These principal characteristics are part of the basis for the containment presumptive remedy suggested by the language in the NCP and set forth by the USEPA (1993b and 1997a) in MSW landfill regulations and guidance (see Section 1.3). Additionally, the presence of a landfill may actually alter the local groundwater flow regime, which can complicate efforts to investigate hydrogeologic and geochemical conditions. Long-term disruption of the natural surface and subsurface soils to place landfilled wastes can result in significant vertical infiltration during extreme precipitation events; these events could then "flush" leachate into lower parts of the aquifer, which—because these zones typically are not exposed to the continuing source of leachate—may be unable to sustain biodegradation processes. Thus, extreme storm events could serve to disperse leachate contamination beyond the acclimated aquifer zones that can control migration through natural attenuation.

Furthermore, the heterogeneity of the landfilled waste, and its resultant leachate constituents, may complicate adequate characterization to support risk analyses, fate and transport calculations, and natural attenuation indicators. The sampling program that may be required to characterize the potential source area and/or complete credible performance monitoring to document large-scale natural attenuation may be costly. Moreover, the chemical characteristics of the leachate may complicate efforts to track common indicators of biodegradation of specific chemical constituents. For instance, chloride is commonly a component of MSW landfills, which may make it difficult to use this compound to infer dechlorination processes in groundwater (Christensen *et al*., 2000).

3.2.2 Evidence for Biotransformation Potential

There is mounting evidence that natural attenuation processes will be—and have been—quite successful at limiting the migration, and eventually the total mass, of leachate plumes containing chlorinated solvents. Christensen *et al*. (1994) state that diverse microbial communities have been identified in leachate plumes. Generally, the most efficient biotransformation of CAHs appear to occur in groundwater that is characterized by an upgradient anaerobic zone and a downgradient aerobic zone (Bouwer, 1994; Carter and Jewell, 1993; Gerritse *et al.,* 1995; Fathepure *et al.,* 1987). This type of geochemical regime is typical beneath MSW landfills undergoing normal stabilization. In the upgradient anaerobic zone, anaerobic reductive dechlorination of highly chlorinated compounds (e.g., PCE and TCE) might result in less-chlorinated daughter compounds (e.g., DCE and VC). These compounds may accumulate in the gas and leachate transport phases, making it possible to track potential natural attenuation processes (Pohland and Kim, 2000). Then, the less-chlorinated compounds could be transformed by direct aerobic oxidation in the aerobic zone (i.e., at the leading-edge or fringe of the plume).

Cometabolic dechlorination may also be an important degradation mechanism for groundwater impacted by chlorinated solvents in MSW landfill leachate. Geochemical conditions are usually sufficient to support the fortuitous transformation of CAHs (e.g., methanogens and sulfate-reducing species are present in impacted environs). However, because of the difficulty in distinguishing between direct and co-metabolic anaerobic reductive dechlorination reactions, the relative roles of these mechanisms are difficult to define and distinguish at the field scale in heterogeneous systems as complex as microbially active landfills.

3.2.3 Case Studies for Natural Attenuation

Initially, simulated landfill studies (Pohland *et al,* 1993; Lesage *et al.,* 1993; Deipser and Stegmann, 1994; Pohland *et al.,* 1998; Oman and Rosqvist, 1999) provided the basis for expecting that CAHs would be biotransformed during the normal phases of landfill stabilization. These simulations indicated that such transformations would be a function of the relative age of a landfill, its construction and operational features, and the temporal and spatial prevalence of the principal reaction mechanisms. Based on the typical acid-formation and methane-fermentation sequences leading to stabilization of the readily degradable organic waste fractions within a MSW landfill, *in situ* direct reductive dechlorination processes are likely to be the dominant degradation mechanism, particularly for the more chlorinated CAHs (e.g., PCE and TCE). This would occur either when other electron acceptors are scarce, as is the case toward the end of the methanogenic phase of landfill stabilization, or when hydrogen is being generated as a consequence of fermentative reactions associated with acid formation from organic substrates.

3.2.3.1 Laboratory Studies

Numerous laboratory batch and field injection *in situ* microcosm experiments have been conducted to investigate the potential for CAH compounds to be biodegraded in MSW landfill leachate (e.g., Reinhart and Pohland, 1991; Nielsen, 1995; Bjerg *et al*.,

1999; Thornton *et al.*, 2000a and 2000b). In laboratory batch studies, the effect of dissolved organic matter in the MSW landfill leachate on the natural attenuation potential of CAHs has been evaluated. Results indicate that chlorinated compounds in leachate with low reducible mineral (e.g., manganese) oxide content and low pH buffering capacity may be less attenuated by natural processes (Thornton *et al.*, 2000a and 2000b). Additionally, the effect of adding water to MSW landfill wastes to promote anaerobic degradation of both chlorinated and petroleum hydrocarbons has been investigated in batch studies (Sanin *et al.,* 2000). Data indicate that the addition of water may increase the bioavailability of organic compounds; therefore, remediation activities designed to reduce moisture infiltration may adversely affect the operation of biodegradation processes.

Field-injection experimental results indicate that PCE, TCE, tetrachloromethane (TeCM), and 1,1,1-TCA can be effectively attenuated in the anaerobic zone immediately adjacent to the landfill (Nielsen *et al.*, 1995). However, once the leachate traveled beyond the immediate area of the landfill, no significant degradation of the CAH compounds was observed. The change in the degradation potential as a function of distance from the source likely is attributable to the effects of the different redox levels.

3.2.3.2 Field Treatability Monitoring

Laboratory studies such as those described in Section 3.2.3.1 quickly gave way to field-scale investigations of the potential for natural attenuation of chlorinated solvents in MSW landfill leachate. A sampling of available field data is summarized below.

- DuPont Necco Park Landfill in New York: Performance monitoring indicates that anaerobic microbial activity is occurring, based on the presence of intermediate breakdown CAH products and metabolic end products. Aerobic, iron-reducing, manganese-reducing, sulfate-reducing, and methanogenic redox conditions have been identified at the site (Lee *et al.*, 1995). The authors report that natural bioremediation is effectively reducing the concentrations of CAHs in groundwater. These processes seem to remain operational even in areas with high pH, high barium concentrations, high salinity, and high concentrations of CAHs. However, the low concentrations of phosphorus (< 1 mg/L), the low groundwater temperatures during the winter, and the availability of electron acceptors and organic substrates in deeper zones may control microbial activity (Lee *et al.*, 1995).

- Wilder's Grove MSW Landfill, North Carolina: This landfill was not constructed with any design features intended to prevent movement of leachate into the groundwater, nor has an engineered cover system been installed. Field performance monitoring indicates that methanogenesis may be the dominant redox process operating beneath the landfill, although iron and manganese reduction may also be occurring. The chlorinated compounds of interest (i.e., TCE and DCE) are being attenuated before significant downgradient migration is observed. Further, it appears that the calcium-carbonate-bearing formations may be enhancing methanogenic biodegradation. This is an important natural enhancement, as leachate from "young" landfills can be acidic (Johnson *et al.*, 1996).

- Former industrial landfill in England: Performance monitoring indicated that reductive dechlorination of highly chlorinated solvents is occurring in the anaerobic environment detected both within the site and at the site boundary. Aerobic reactions predominate farther downgradient. Mathematical modeling analyses suggest that the CAH plume poses no immediate threat to potential receptors (Ellis *et al.,* 1999).

- North Bay Municipal Landfill, Ontario, Canada: Even under the influence of rapid groundwater flow velocities (i.e., 20 to 170 meters per year, averaging 75 meters per year), the CAH plume originating from the landfill is naturally restricted to the immediate vicinity of the landfill. Two of the primary CAH compounds, TCE and 1,1,1-TCA, seem highly susceptible to biodegradation processes under the conditions of active methane generation (Barker *et al.,* 1986).

- Sengeloese Landfill in Denmark: This landfill is characterized by stabilized anaerobic conditions with low methane production.. Although site data indicate that PCE is not being transformed under these conditions, other chlorinated compounds (i.e., 1,1,1-TCA and TeCM) are being effectively attenuated (possibly by abiotic processes) (Kromann and Christensen, 1998).

- Closed municipal landfill in fractured bedrock: CAH compounds are being reductively dechlorinated, as indicated by the presence of daughter products and abundant chloride ions. The predominant biodegradation processes include iron- and sulfate-reduction. Fate and transport modeling indicate that the contaminant plume will not extend beyond 100 feet from the landfill. As in many landfills, the elevated concentration of dissolved organic solutes in the leachate appear to provide a rich substrate for the microorganisms that participate in the degradation of chemicals in the leachate (Brasaemle *et al.,* 1997).

Clearly, there is an evolving body of knowledge with regard to the potential for chlorinated solvents to naturally attenuate in groundwater impacted by MSW landfill leachate. In many cases, this attenuation potential has been suggested merely by the continued absence of CAHs in wells downgradient from known release sites (IT Corporation, 2000). Yet, the potential for natural attenuation of CAHs is by no means universal or even presumed. Case studies repeatedly demonstrate that there are no general characteristics that can be identified to easily and correctly predict the potential for natural attenuation of CAHs at any particular site (Parsons, 1999, Reinhard *et al*, 1994). Consequently, the integration of natural attenuation processes into the recognized management strategy for MSW landfills will need to be completed on a site-by-site basis.

3.2.4 Enhancing *In Situ* Biodegradation

In addition to relying on naturally occurring processes, it may be possible to engineer controlled degradation processes for chlorinated solvents. Thus, rather than emphasizing performance monitoring, these approaches would incorporate an engineered treatment component. This treatment component at MSW landfills, either lined or unlined, usually would involve leachate (groundwater) collection and recirculation. The collected leachate (groundwater) could be amended before reinjection to promote more rapid

and/or complete *in situ* transformation reactions. As noted by USEPA (1999c), these types of treatment trains are not strictly included in their definition of natural attenuation.

In situ biorestoration schemes may focus on stimulating aerobic biodegradation in naturally anaerobic zones to ensure complete mineralization of chlorinated compounds (Berwanger and Barker, 1988). *In situ* biorestoration may also seek to promote and establish dual treatment zones to maximize CAH degradation (Lesage *et al.*, 1993). For example, at a Superfund landfill in New England, sequential anaerobic and aerobic treatment zones are being supplemented with subsurface amendments. Specifically, an electron donor (sodium benzoate) is being injected in the anaerobic zone at the toe of the landfill to enhance the biodegradation of CAHs; then, oxygen is injected downgradient to create an aerobic treatment zone to enhance biodegradation of other compounds (e.g., petroleum hydrocarbons and VC) and to stimulate the precipitation of arsenic (Turpie *et al.*, 2000). Such an engineered remedy provides for continued infiltration through landfilled wastes, so that existing microbial populations are preserved and nourished.

3.3 ALTERNATIVE LANDFILL DESIGN CONCEPTS

Pohland (1980) reports that the concept of leachate containment, collection, and recycling using the landfill as a biological and physical/chemical treatment system was conceived and tested at Georgia Tech during the 1970s. This reference presents selected results of early recirculation studies, and demonstrates that leachate recycling can be a viable management option, particularly in cases where leachate production is inevitable. Townsend *et al*. (1996) state that relatively slow landfill stabilization rates associated with single-pass landfill designs led to the concept of leachate recirculation technology and the construction of test systems to observe the effects of leachate recycle on landfill stabilization.

As of 1993, few comprehensive reports of leachate recycling as an *in situ* treatment option in the U.S. had been published (Pohland *et al*., 1993). As of 1996, the most detailed leachate recycling investigations had been conducted in laboratory-scale experiments, landfill lysimeters, and controlled landfill cells. Investigations of full-scale leachate recycling operations had been reported, but the details of investigation did not approach that of smaller-scale studies (Townsend *et al*., 1996).

A useful description of the general principles and effects of bioreactor landfills is presented by Pohland and Kim (2000). Pohland and Harper (1995) describe a near-full-scale bioreactor demonstration project in Mountain View, California, and state that information is also available concerning several field-scale landfills in Germany where leachate recycling is being used. Investigations conducted by USEPA (1988) indicated that more than 200 MSW landfills were employing leachate recirculation as a leachate management method. Magnuson (1998) conducted a review of current literature and interviewed operators and researchers currently involved with leachate recirculation in MSW landfills in order to determine the state of the practice as well as perceived advantages and disadvantages.

It should be noted that the bioreactor concept as discussed in the literature is primarily applied to newer landfills that are intentionally engineered to operate in this fashion. Retrofitting of older landfills to operate as bioreactors is not discussed as extensively.

However, a useful description of design elements associated with implementing leachate recirculation at older landfills that do not have bottom liners is provided by Harper and Just (1995).

The benefits of leachate recirculation typically cited in the literature include more rapid decomposition of organic waste and associated landfill stabilization, with a concomitant reduction in contaminant mass discharged to the environment, and a reduction in the overall cost of leachate treatment (Onay and Pohland, 1998; Pohland and Kim, 2000; Townsend *et al*., 1996; Tittlebaum, 1982; Miller and Emge; 1997; Magnuson, 1998; Barlaz and Gabr, 2000; Bendz *et al*., 1997). Waste settlement that accompanies the stabilization process is discussed by Gabr *et al*. (2000) and El-Fadel (1999).

Full-scale application of leachate recirculation has been hampered by an inability to uniformly apply leachate to the mass waste. Much of this problem is due to the heterogeneous nature of the disposed waste, which promotes preferential routing and short-circuiting of leachate flows (Johnson *et al*, 1998; McCreanor and Reinhart, 2000; Townsend *et al*., 1995; Oman and Rosqvist, 1999).

Addition of various amendments to the subsurface, either via leachate recirculation or direct injection, has been performed with varying degrees of success, primarily in batch- or pilot-scale tests. The objective of amendment injection has been to create subsurface aerobic and/or anaerobic treatment zones in or adjacent to landfills within which microbial degradation of contaminants is stimulated. References that describe such activities (and the amendment[s] used) include Young *et al*. (1997) and Brigmon *et al*. (1999) (air, nitrous oxide, triethylphosphate, methane); Tittlebaum (1982) (pH and nutrient control); Berwanger and Barker (1988) (hydrogen peroxide); Barr *et al*. (1997) (air); and Turpie *et al*. (2000) (sodium benzoate and oxygen).

3.4 ALTERNATIVE COVER DESIGNS

Several alternative cover designs have emerged in the past few years that would support treatment-based MSW landfill management strategies. These cover designs range from slightly modified standard RCRA MSW landfill covers, to vegetative (ET) covers, to basic control covers that may prevent direct receptor exposures yet provide for controlled moisture application. This subsection briefly summarizes recent information on alternative cover designs that might be compatible with promoting *in situ* natural attenuation processes for CAHs. Weand *et al.* (1999) and Suthersan (2001) provide comprehensive reviews of state-of-the-art cover design options.

3.4.1 Capillary-Barrier Designs

Capillary-barrier covers consist of a relatively fine-grained soil layer overlying a relatively coarse-grained layer. The barrier is created by the large change in pore sizes between the two layers. Capillary force causes the layer of fine soil overlying the coarser layer to hold more water than if there were no change in particle size between the layers (Hauser *et al*., 2001a). Hill AFB, Utah explored the potential to use capillary barrier covers to control the infiltration of natural precipitation (Warren *et al.*, 1996a). The capillary-barrier designs investigated consisted of 150 centimeters (cm) of topsoil over 30 cm of gravel. An additional layer of 1-cm gravel/ mulch covering was included to

control erosion. The cover was seeded with a mixture of native perennial grasses, and planted with two shrub species (rabbitbrush and saltbrush).

Monitoring results indicated that the capillary-barrier type cover allowed between 12 and 15 percent of the precipitation to infiltrate. In comparison, a simple exposure-control cover allowed 20 percent of the precipitation to infiltrate. However, the capillary-barrier cover was unable to shed water laterally along the capillary break, especially during periods of rapid wetting. However, in arid environments, these types of covers may limit infiltration by allowing ET processes to dominate.

3.4.2 Evapotranspiration Cover Designs

The ET landfill cover concept has been clearly defined (e.g., Hauser and Shaw, 1994a and 1994b; Hauser *et al.*, 1994, 1995, 1996; Hauser, 1997; Weand and Hauser, 1997, Hauser and Weand, 1998; Suthersan, 2001). ET covers are designed to minimize movement of precipitation into underlying landfilled wastes by using common phyto-processes and the soil's water-holding capacity. These types of alternative covers may play an important role in promoting natural attenuation within landfill environments because they:

- Enhance natural biodegradation processes;
- Allow for passive venting of gaseous byproducts of biodegradation and allow oxygen to migrate into the landfill, thereby facilitating aerobic biodegradation;
- Provide a "green" ecosystem that is an attractive alternative to a RCRA cap; and
- Can be installed at lower cost and less risk to public safety than a RCRA cap, and, once the cover is established, has a natural stability that minimizes long-term maintenance requirements (Hauser *et al.*, 2001a).

However, it is important to note that, although the regulatory climate may be evolving toward more frequent use of ET covers, some constructed vegetative covers have not met the requirements for an effective landfill cover (e.g., provided for equivalent "dry entombment"). Failures have generally been attributed to overcompaction and insufficient soil thickness to store infiltrating precipitation (normal and/or extreme events) and support healthy plant growth (Warren *et al.*, 1996b; Anderson and Lovley, 1997).

3.4.3 Organic Sub-Layered Covers

Haas *et al.* (2000) describe the effects on degradation of CAHs of a 30-foot by 15-foot surface amendment plot at Offutt AFB, Nebraska, constructed with an 18-inch layer of mulch (shredded tree compost) and sand mix. The plot was bermed to promote downward delivery of organic matter via infiltration to TCE-contaminated groundwater. Subsequently, the *cis*-1,2-DCE/TCE ratio in shallow groundwater beneath the test plot area varied from 0.024 ugradient from the plot to 0.35 downgradient, with a mean increase in the ratio of a factor of 15. These data indicate that the organic mulch layer enhanced anaerobic, reducing conditions that are conducive to the reductive dechlorination of TCE. Similar benefits were obtained at the same site by installing a

vertical, subsurface PRB filled with a mixture of mulch and sand. This concept could be extended to landfills by constructing an ET cover that incorporates an organic mulch sub-layer.

SECTION 4

ALTERNATIVE CLOSURE DESIGNS

Landfilled wastes can be complex mixtures consisting of multiple types of contaminants. However, the focus of this report is on the management, remediation, and closure of military landfills at which the predominant COCs requiring remediation are CAHs. To guide DoD remedial project managers (RPMs) in the selection of an appropriate remedial strategy for CAH-contaminated landfill(s), three decision trees were developed. These decision trees, which should be followed sequentially, guide the user through the following general remedy-selection steps:

- Landfill Screening;
- Detailed Remedial Alternative Assessment; and
- Landfill Cover Designs.

The decision trees and related topics are discussed in detail in subsections 4.1 through 4.3.

Boyer *et al*. (1999) present a series of 15 detailed decision trees pertaining to remediation of USAF MSW landfills; however, leachate-recirculation and landfill-bioreactor strategies were not included as options in these decision trees. A primary focus of the Boyer *et al*. (1999) decision trees was assessment of the appropriateness of the USAF's (1997) *Presumptive Remedy for Landfills*, which includes implementation of some or all of the following five components:

- Landfill cover;
- Landfill gas collection and treatment system;
- Source-area groundwater-control system;
- Leachate collection and treatment system; and
- Institutional controls.

Boyer *et al*. (1999) note that selected presumptive-remedy components are often combined with other actions to form a complete remedy. Therefore, consideration of other remedial strategies, such as bioreactor-landfill approaches included in the decision

trees developed during this evaluation, can be viewed as supplemental to the presumptive remedy strategies described by Boyer *et al.* (1999).

4.1 INITIAL LANDFILL SCREENING

The purpose of the Landfill Screening Decision Tree is to determine whether engineered remediation of CAHs in groundwater is required, or whether cleanup goals can be met within an acceptable time frame via natural attenuation. The Landfill Screening Decision Tree is presented on Figure 4.1. Elements that form the basis of the decision logic are reviewed in the following subsections.

4.1.1 Presence of CAH Contamination

If the nature and/or extent of groundwater and/or surface water contamination at or downgradient from the landfill has not been characterized, then a site assessment employing a records search and environmental sampling should be performed. If groundwater and/or surface water have not been impacted by undesirable contaminants, the RPM should ensure that soil-gas and groundwater monitoring programs are in place to detect any future releases from waste containers that may be present in the landfilled materials. Soil-gas monitoring is recommended to provide early warning of released volatile contaminants that may not yet have migrated to the water table or into deeper parts of the affected aquifer. Soil-gas samples can be obtained via one or more of the following methods:

- From gas-collection/vent wells installed in the landfill;
- From dedicated soil-gas monitoring wells screened in the vadose zone within or beneath the landfilled wastes;
- Using passive soil-gas samplers (e.g., sorbent material) buried beneath the landfill cover; or
- Using hollow, stainless-steel rods driven beneath the landfill cover.

If site assessment results indicate that significant groundwater contamination is not present, then it is unlikely that a final cap consisting of a low-permeability cover is required. Given the ages of most DoD landfills (> 20 years), the potential for contaminant "time bombs," consisting of intact containers that could deteriorate and release large quantities of contaminants, generally is low. Therefore, there may be little justification for constructing expensive, barrier-type covers to entomb the waste in a relatively dry state. Alternative cover options (e.g., ET covers) that would prevent direct contact with landfilled materials while allowing controlled percolation of precipitation into the landfill should be evaluated (see Section 4.3). Moreover, in arid or semi-arid climates, the percolation-control properties of a properly designed ET cover may be comparable to those of a conventional low-permeability cover. In some cases, maintenance and, if necessary, enhancement of an existing soil cover may be sufficient to prevent erosion and promote healthy vegetation.

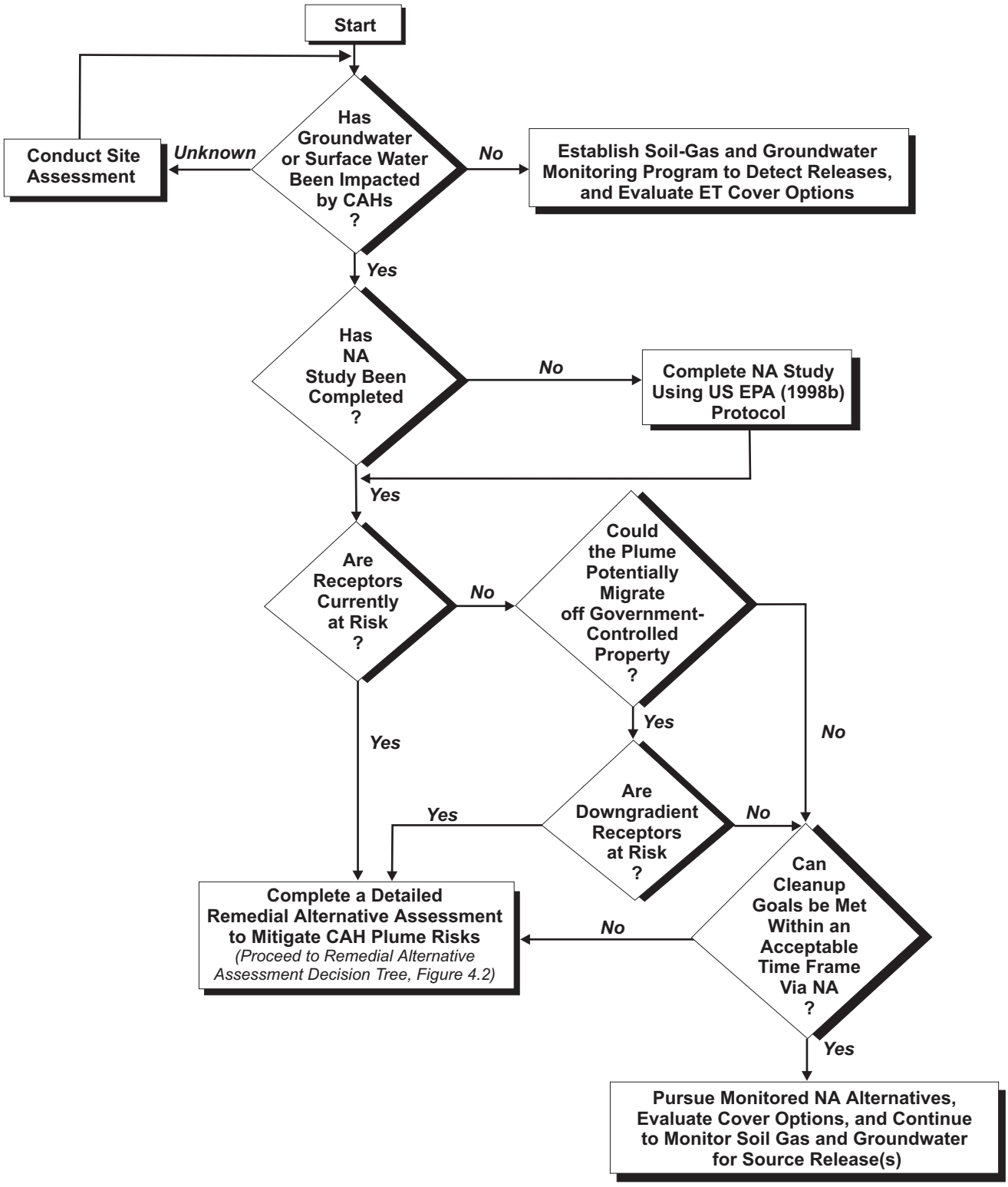

FIGURE 4.1

LANDFILL SCREENING DECISION TREE

Military Landfill Closure
Design Evaluation

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Notes:
CAHs = Chlorinated aliphatic hydrocarbons
ET = Evapotranspiration
NA = Natural Attenuation

4.1.2 Natural Attenuation Evaluation

Once any landfill impacts on groundwater and/or surface water have been quantified, a natural-attenuation treatability study (TS) should be performed to assess whether cleanup goals can be met within an acceptable time frame via natural attenuation, or whether engineered remediation is required. As described in Section 3.2, numerous studies have shown that geochemical conditions conducive to natural biodegradation of CAHs are present in groundwater beneath or immediately downgradient from landfills. When natural attenuation processes (i.e., biodegradation, dispersion, sorption, and volatilization) are shown to be capable of attaining site-specific remediation objectives in a time period that is reasonable compared to other alternatives, MNA may be selected, alone or in combination with other more active technologies and institutional controls, as the preferred remedial alternative (USEPA, 1998b). A technical protocol for data collection and analysis to evaluate MNA for remediating groundwater contaminated with CAHs has been published by USEPA (1998b).

The natural attenuation TS should include a contaminant fate and transport analysis for COCs, in which the effects of biodegradation are accounted for, to assess the future migration and persistence of dissolved CAHs. This type of evaluation can be used to evaluate the degree to which the contamination poses future risks to potential downgradient receptors. The fate and transport analysis typically is performed using analytical or numerical models.

4.1.3 Risk Analysis

Based on the results of the site assessment and the natural attenuation TS (Sections 4.1.1 and 4.1.2), risks posed by the COCs to human and or ecological receptors can be evaluated for current and expected future land-use scenarios and contaminant plume configurations. For a COC to pose a risk, four elements must exist at a site:

- A source of chemical contamination that exceeds or could generate chemical contamination above health-protective or aesthetic standards;
- A mechanism of contaminant release;
- An exposure-point at which human or ecological receptors could contact the chemical; and
- A completed exposure route (e.g., ingestion or inhalation) through which the receptor will contact the chemical.

If any one of these four elements is absent at a site, there is no current risk. The reduction or elimination of risk can be accomplished by limiting or removing any one of these four elements from the site.

If contaminant fate and transport analyses indicate the potential for a CAH plume to migrate off DoD-controlled property, and downgradient receptors are potentially at risk, then a detailed remedial alternatives assessment should be completed to select an appropriate remedy that will mitigate risks posed by the plume. Remedial alternatives that are recommended for consideration at DoD landfill sites are discussed in detail in

Section 4.2. Regardless of whether or not the CAH plume has the potential to migrate off government-controlled property, if the plume does not pose a risk to downgradient receptors, then completion of a detailed remedial alternative assessment may not be required.

As shown on Figure 4.1, the key question to be answered is, "Can cleanup goals be met within an acceptable time frame via natural attenuation without engineered intervention?" This question can be evaluated using calculated contaminant decay rates derived during the natural attenuation TS (USEPA, 1998b), usually in conjunction with numerical or analytical fate and transport models. If cleanup goals can be met within an acceptable time frame via natural attenuation, then a remedial alternative for groundwater consisting of MNA should be pursued. It is important to note that the time frame for many groundwater remediation scenarios, both passive and engineered, will be determined by the slow release of contaminants from the source. In such cases, the time frames to achieve final remediation goals via MNA will be essentially the same as for more "active" groundwater pumping scenarios.

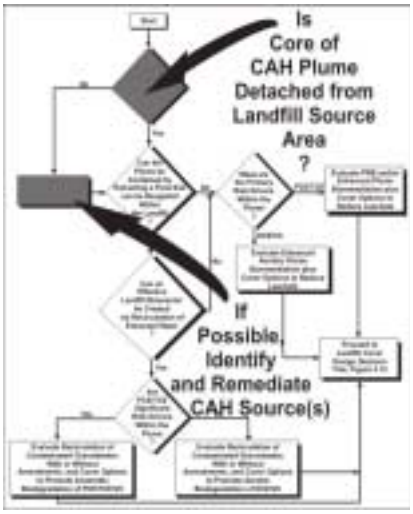
In addition to MNA for groundwater, a suitable landfill cover option should be identified as a source-control measure, as discussed in Section 4.3. Continuing or new releases of contaminants within the landfill can substantially increase the migration potential and/or persistence of the CAH plume, and may trigger reevaluation of the remedial approach (e.g., from MNA to a more aggressive, engineered alternative). Therefore, continued monitoring of groundwater and soil gas is a key component of any MNA remedy at MSW landfill sites.

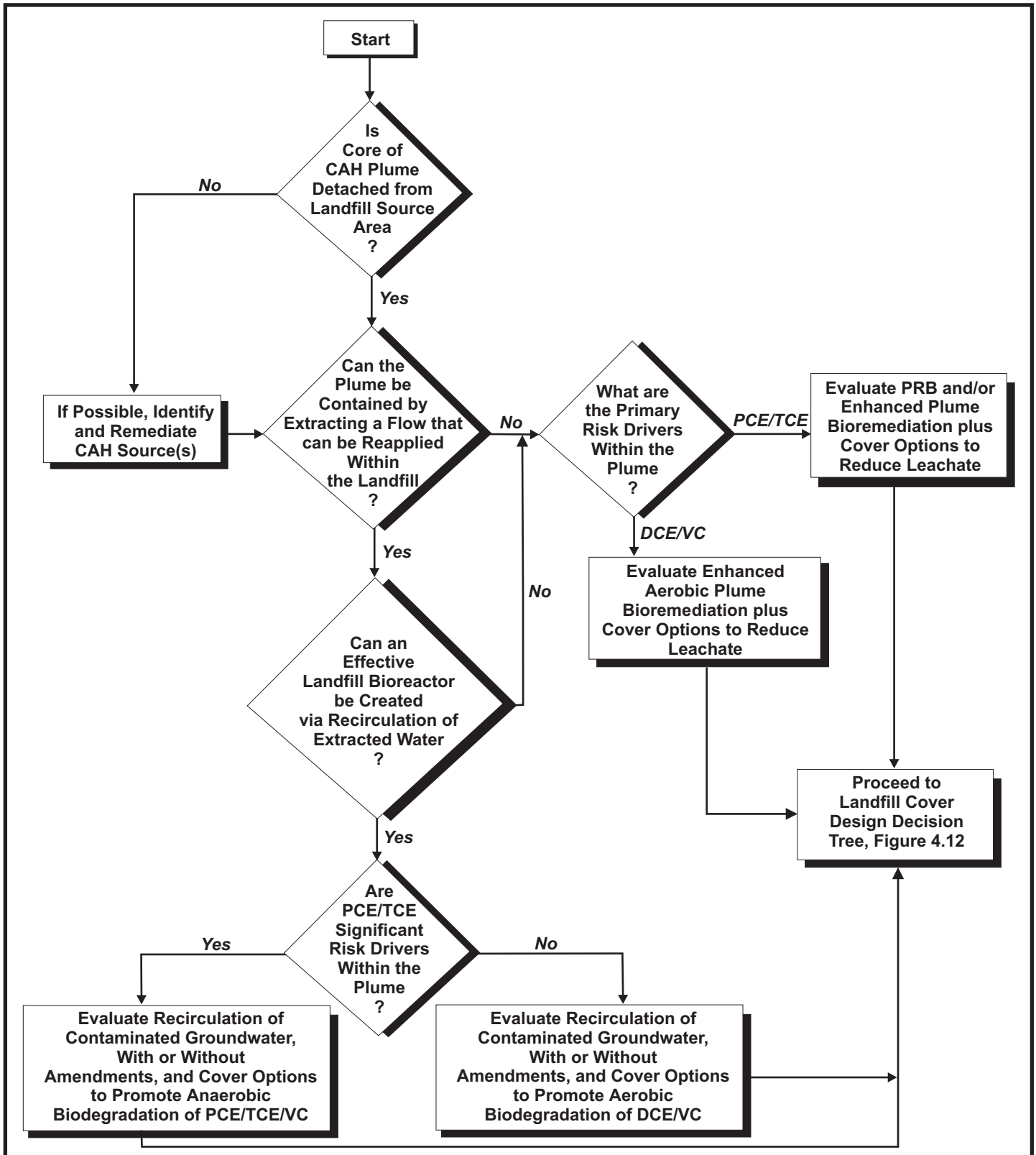
4.2 REMEDIAL ALTERNATIVES ASSESSMENT

The Remedial Alternatives Assessment decision tree is depicted on Figure 4.2. The purpose of this decision tree is to guide the user through the process of selecting an appropriate remedial alternative when risk or time considerations dictate a more aggressive approach than MNA, and to describe the conditions under which operation of the landfill as a bioreactor with collection and recirculation of leachate-contaminated groundwater may be an appropriate (and recommended) alternative.

4.2.1 Source Identification and Removal

Commonly implemented options for controlling the effects of landfill leachate generation, such as extraction and treatment of contaminated groundwater, do not address the source of contamination. Buried wastes can act as continuing sources of groundwater contamination for decades; therefore, source removal, if feasible, can dramatically reduce remediation time frames and long-term cleanup and monitoring costs. Leaking containers should be identified and removed whenever possible, because these sources would be unaffected by any form of *in situ* treatment or air/water extraction.





Notes:
CAH = Chlorinated aliphatic hydrocarbons
Core = Center of plume mass
DCE = Dichloroethene
PCE = Tetrachloroethene
PRB = Permeable reactive barrier
TCE = Trichloroethene
VC = Vinyl Chloride

FIGURE 4.2

DETAILED REMEDIAL ALTERNATIVE ASSESSMENT DECISION TREE

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Design Evaluation

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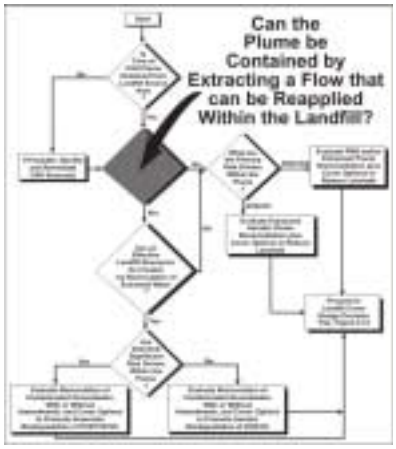
If the core, or center of mass, of a CAH plume in groundwater is detached (i.e., is located downgradient) from the landfill source, as shown on Figure 4.3, it can be inferred that the source of contamination in the landfill has been, or is becoming, depleted. In such cases, source removal/reduction may not be required. Conversely, if the center of mass in the plume extends from or is localized beneath the source area, then an active, continuing source likely is present, and reasonable efforts to locate and remove/remediate significant sources may provide significant payback in terms of reduced plume migration and persistence.

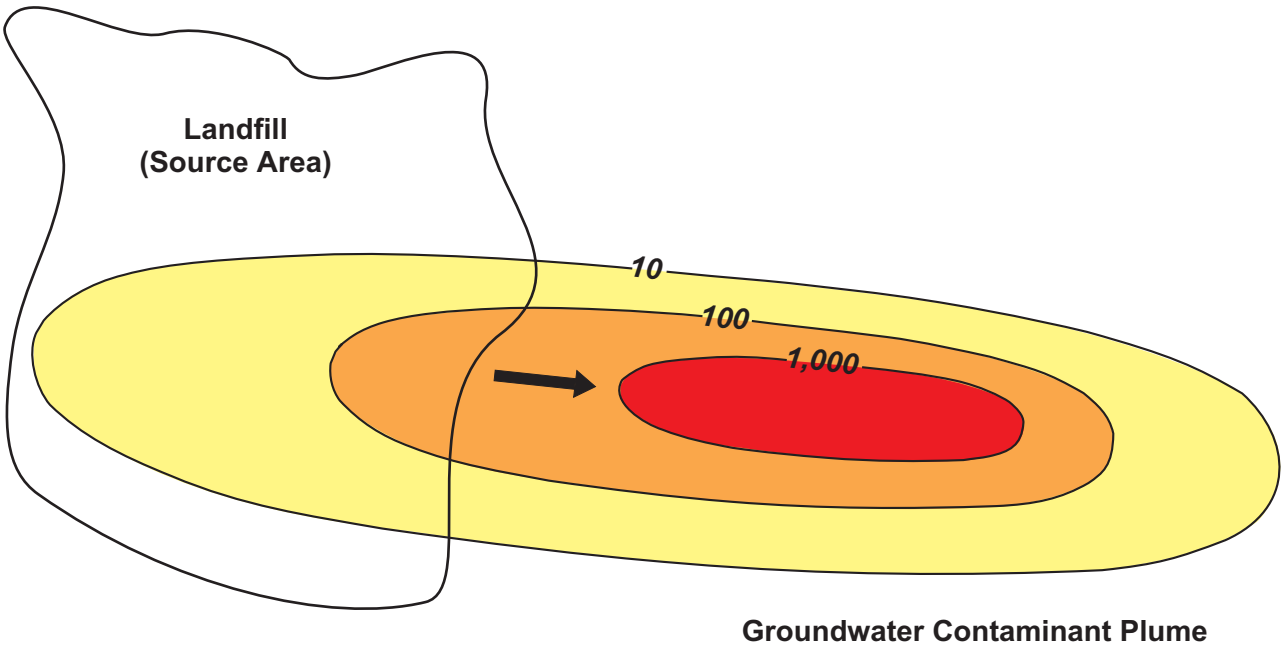
Potentially useful landfill source-location methods include review of historical aerial photographs, geophysical (e.g., magnetic or electromagnetic) surveys, and soil-gas sampling. Once a potentially significant source area is located and adequately characterized (e.g., using test pits or soil borings), it can be excavated and removed (if the volume of wastes is small) or remediated *in situ.* Potentially applicable *in situ* technologies include conventional or thermally enhanced soil-vapor extraction (SVE), redox alteration, enhanced biodegradation, or (for dense nonaqueous-phase liquid [DNAPL] removal) surfactant flushing. The *Remediation Technologies Screening Matrix and Reference Guide* (http://www.frtr.gov/matrix2) is a useful compendium of source-removal technologies. Boyer *et al*. (1999) suggest that, as a general rule of thumb, hotspots less than 100,000 cubic yards (yd^3) in volume may warrant consideration for excavation or treatment. However, this size criterion should not be considered a rigid threshold because remedy selection also depends on other factors (e.g., the depth of the buried source and the availability of aboveground treatment/disposal options).

4.2.2 Determining the Feasibility of Plume Capture and Recirculation



A remedial-alternatives assessment for a DoD landfill should include an evaluation of the potential benefits of capturing leachate-contaminated groundwater and recirculating it into the landfill (i.e., operating the landfill as a bioreactor). The capacity of the landfill to assimilate recirculated groundwater will be limited, and will depend on the areal extent and depth of landfilled materials, the field capacity of landfilled materials (defined as the threshold moisture content above which leachate generation occurs), the recirculation system design (e.g., recirculation wells, trenches, or ponds), and the optimal moisture content required to operate an effective bioreactor. While optimal *in situ* bioremediation of CAHs occurs in moist environments, an overabundance of water can promote excessive leaching, cause leachate breakout seeps at the ground surface, stimulate migration of the dissolved contaminant plume in groundwater, and alter geochemical conditions in the subsurface such that microbial processes that degrade highly chlorinated solvents (e.g., anaerobic reductive dechlorination) may be compromised.

In some cases, the volumetric flux of leachate-contaminated groundwater migrating away from a landfill may exceed the recirculation capacity of the landfill. This situation is most likely to occur when the affected aquifer has a relatively high transmissivity (i.e.,



Groundwater Contaminant Plume

Legend

1,000 — Contaminant Concentration

Groundwater Flow Direction

Landfill Boundary

Not to Scale

FIGURE 4.3

DISSOLVED PLUME WITH DETACHED CENTER OF MASS

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Design Evaluation

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has moderate to high hydraulic conductivity, and/or is relatively thick). In this case, implementation of recirculation and bioreactor strategies may not be feasible. Alternatively, the most contaminated portion of the groundwater plume could be targeted for extraction and recirculation to the landfill, and the remaining volume could be allowed to attenuate naturally *in situ* or be treated *ex situ* and disposed of in another manner. If capture and treatment of leachate-contaminated groundwater is already being implemented or is part of an approved remedy, recirculation of a portion of the groundwater into the landfill may allow a reduction in treatment and disposal costs. Recirculation also will accelerate the source-weathering process, and under optimum conditions, will degrade the CAH source via reductive dechlorination and other biological and/or abiotic pathways. General guidance regarding landfill characteristics that would be conducive to recirculation of leachate-contaminated groundwater (and vice versa) is provided in Table 4.1.

4.2.3 No-Recirculation Remedial Alternatives

If recirculation of leachate-contaminated groundwater is not feasible for any of the reasons listed in Section 4.2.2, and MNA is not a viable remedial alternative (e.g., due to imminent risks posed to potential receptors by the plume), then alternative remedial options must be evaluated. The nature of these options will depend primarily on plume chemistry (i.e., what are the primary risk drivers within the plume?) and hydrogeologic conditions.

Consideration of relatively low-cost, *in situ* remedial approaches in addition to conventional groundwater extraction (GWE) and *ex situ* treatment is recommended because, while most GWE systems can remove contaminants that are dissolved in the groundwater, they are limited in their ability to remove residual nonaqueous-phase liquid (NAPL) contaminants that are sorbed to or trapped within the aquifer matrix (i.e., their effectiveness is diffusion limited). Diffusion-limited removal rates are the most common shortcoming of all *in situ* treatment methods. If little or no NAPL is present, and site soils are sufficiently permeable to air and/or groundwater flow, then extraction technologies often can achieve cleanup goals within a reasonable time frame. However, at sites where residual NAPL is trapped within the soil matrix, or if the aquifer contains a significant fraction of low-permeability (fine-grained) sediments, slow rates of contaminant diffusion from NAPL into soil gas or groundwater may result in unacceptably long cleanup times.

4.2.3.1 No-Recirculation Remedial Alternatives for Less-Reduced Solvents

If *in situ* natural biodegradation of the parent solvents (e.g., PCE and TCE) has been relatively ineffective, and these less-reduced (i.e., highly-chlorinated) compounds are the primary groundwater COCs, then use of PRB technology and/or enhanced plume bioremediation should be considered. Under these circumstances, landfill cover options to reduce leachate generation also should be considered for plume remediation. Cover options are discussed in Section 4.3.

TABLE 4.1
SITE CHARACTERISTICS INFLUENCING BIOREACTOR OPERATION AND EFFECTIVENESS
MILITARY LANDFILL CLOSURE DESIGN EVALUATION

Characteristic	Comments
Subsurface Heterogeneity	Absence of clay layers within landfilled wastes promotes uniform percolation of recirculated water, prevents preferential flow and short-circuiting, and maximizes treatment effectiveness.
Hydraulic Conductivity	Higher vertical hydraulic conductivities (of landfilled wastes) promote infiltration of recirculated water. Higher horizontal hydraulic conductivities (of the aquifer surrounding the landfilled wastes) reduce groundwater mounding and accompanying radial plume spreading.
Impacted Aquifer Thickness	Capture of contaminated groundwater in thin aquifers would be more effective than in thick aquifers. Plume interception in thin surficial aquifers would be relatively simple using collection trenches, and would generate smaller flows for recirculation.
Hydrogeologic Complexity	Relatively simple hydrogeologic settings (e.g., more isotropic and homogeneous) promote effective and potentially more cost-effective plume capture and facilitate accurate prediction of the aquifer's response to recirculation of leachate-contaminated groundwater. As a result, the landfill bioreactor can be operated as a closed-loop system.
Contaminant Source Area Definition	Knowledge of contaminant source area locations and characteristics promotes enhanced leaching of contaminants and targeted construction of subsurface treatment zones, and speeds landfill stabilization.
Chemical and Geochemical Characterization	Defining the nature of chemicals of concern and subsurface geochemical conditions facilitates informed decision-making regarding the need to amend recirculated water (e.g., with electron donors/acceptors) to promote desired subsurface geochemical conditions.
Recirculation Rate and Landfill Field Capacity	A balance should exist between the groundwater extraction rate necessary to achieve the desired plume capture and the ability of the landfill to accept the extracted leachate-contaminated groundwater without excessive groundwater mounding and formation of surface seeps. Large landfills (in terms of area and thickness above the water table) typically are conducive to higher recirculation flow rates and vice versa. Landfilled wastes comprised primarily of construction debris may have a lower field capacity than municipal solid waste.

Permeable Reactive Barriers. PRB technology is a low-cost, low-maintenance, passive *in situ* method for treatment of CAH-contaminated groundwater. Reactive material is placed in the subsurface in the natural-gradient flow path of a dissolved CAH plume. The contaminants are removed as the plume flows through the PRB (Figure 4.4). Depending on site-specific conditions, various barrier configurations can be employed, including continuous reactive walls (i.e., trenches filled with permeable, reactive media), funnel-and-gate systems, *in situ* treatment vessels, GeoSiphon™ cells, and aboveground reactors. The vast majority of PRBs installed to date consist of excavated trenches filled with reactive material. However, installation of PRBs below traditional excavation depth limits is being investigated. For example, Hocking *et al*. (2000) describes the installation of PRBs to depths greater than 120 feet.

The majority of PRBs installed to date use zero-valent iron as the reactive medium, though other reactive materials such as bark mulch (Haas *et al*., 2000) are being investigated. Zero-valent iron has the ability to reductively dehalogenate CAHs such as TCE to ethene. This technology has been successfully implemented at over 50 sites in North America and Europe. An exclusive patent on the use of this technology is held by Envirometal Technologies, Inc. of Waterloo, Ontario, Canada. This firm generally performs bench-scale column testing to determine PRB design parameters for a particular site. Detailed testing of this type is recommended to gauge the effect, if any, of site-specific constituents on iron-CAH process chemistry.

When properly designed and implemented, PRBs are capable of remediating a number of contaminants, including many common CAHs, to regulatory numerical groundwater standards. It is currently believed that these systems, once installed, will have extremely low maintenance costs for at least 5 to 10 years (USEPA, 1998b).

Enhanced Bioremediation. Engineered bioremediation technologies are used to enhance the natural *in situ* processes that degrade CAHs in contaminated soil and groundwater. Technologies include bioaugmentation and addition of nutrients, electron donors, and electron acceptors (such as oxygen) to the subsuburface. A summary of engineered approaches to *in situ* bioremediation is provided by USEPA (2000e).

For highly chlorinated solvents such as TCE, addition of electron donors to the subsurface to promote anaerobic reductive dechlorination is one of the most common approaches. Figure 4.5 illustrates the creation of a subsurface reactive zone via amendment injection into the aquifer. Electron donors are carbon sources, such as propane, methane, molasses, or vegetable oil, that provide a food source (substrate) for microbes and yield hydrogen as they are fermented. As described in Section 2.1, hydrogen supplies electrons that are used by microorganisms in the transformation of a wide range of chlorinated hydrocarbons. The USAF currently is preparing a protocol document for enhanced reductive dechlorination via vegetable-oil addition. Direct hydrogen addition, wherein hydrogen is delivered without the use of a fermentation substrate or carbon source, is another *in situ* bioremediation technology currently under development for chlorinated solvent plumes (Newell *et al*, 2000). Hydrogen-Release Compound (HRC™) is a commercially available product for delivering electrons in a timed-release fashion.

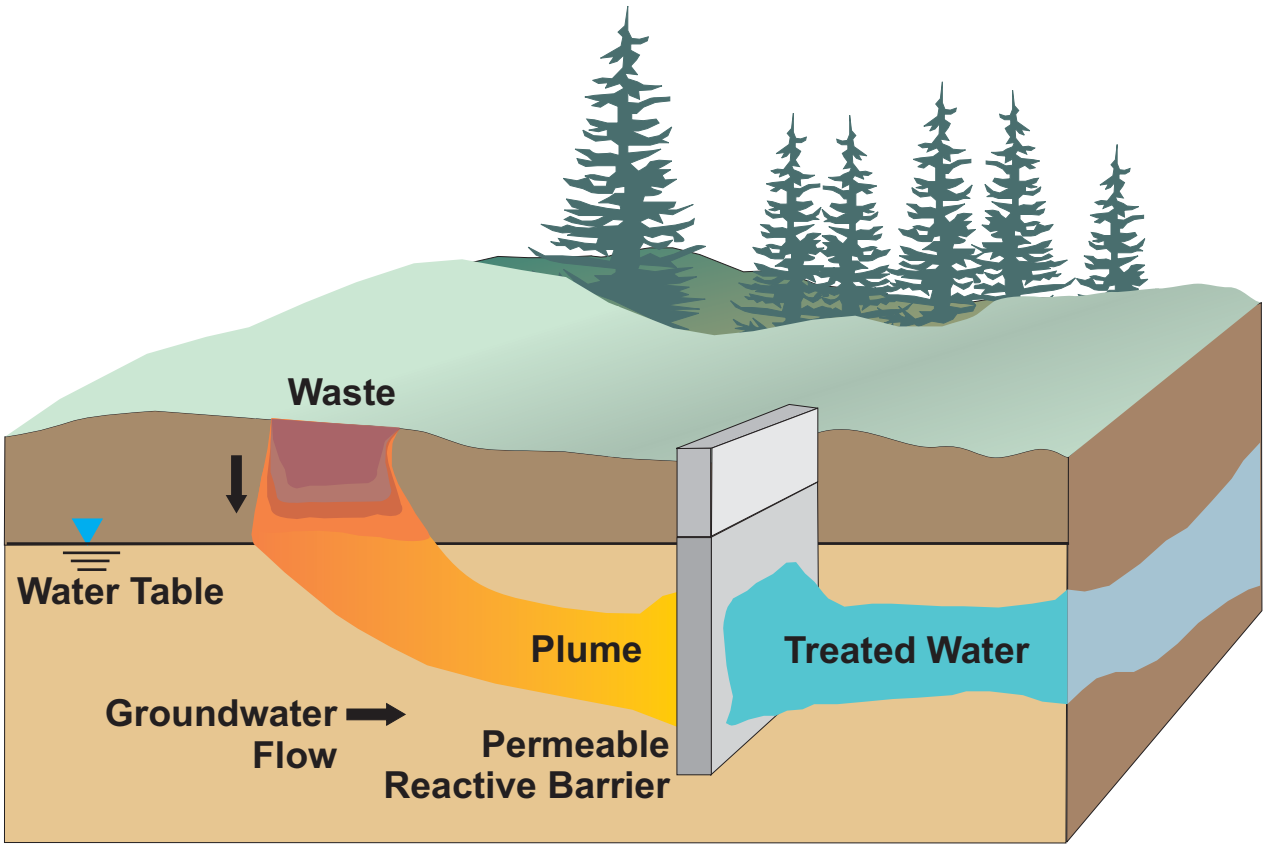

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FIGURE 4.4

PERMEABLE REACTIVE BARRIER

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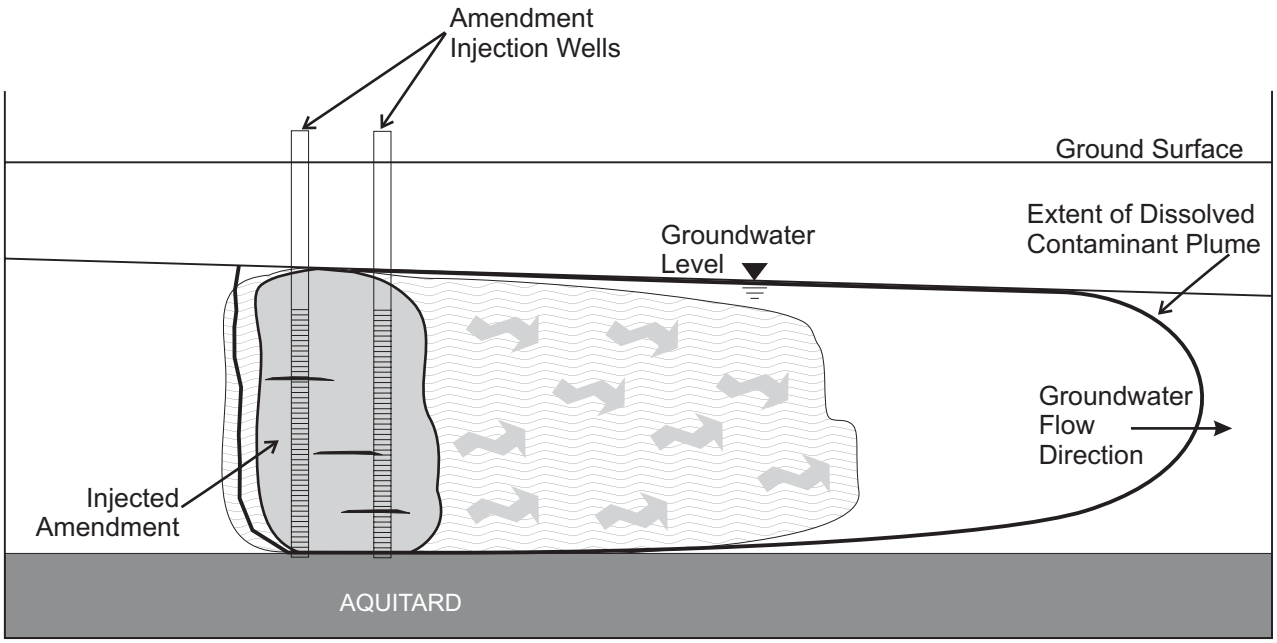

Legend

Organic Amendment Zone

Reactive Zone: Area of Enhanced Contaminant Degradation

DNAPL

Not to Scale

FIGURE 4.5

AMENDMENT INJECTION TO CREATE A REACTIVE ZONE

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Direct Chemical Oxidation. Direct chemical oxidation involves the injection of strong oxidants, such as hydrogen peroxide, Fenton's Reagent (hydrogen peroxide plus an iron catalyst), potassium permanganate, or ozone, to contaminated soil, sludge, and groundwater. Chemicals successfully targeted for treatment using this approach have included CAHs, semivolatile organic compounds (SVOCs), polychlorinated biphenyls (PCBs), and pesticides (USEPA, 1998b). Direct chemical oxidation is rapid and generally results in innocuous byproducts. Due to its relatively high cost, this technology has been most frequently applied at sites with small source areas with high concentrations of CAHs, including DNAPL areas. At CAH-contaminated MSW landfills, direct chemical oxidation should be evaluated along with other source-reduction methods such as excavation and enhanced bioreactor designs.

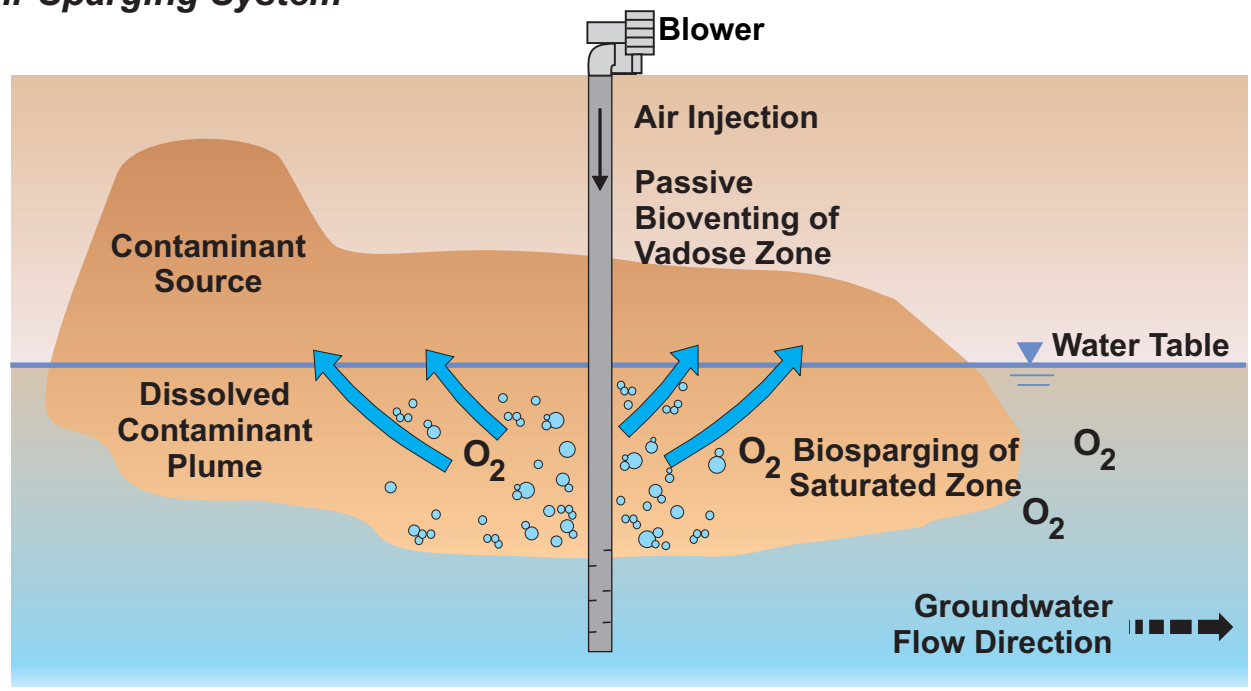
4.2.3.2 No-Recirculation Remedial Alternatives for More-Reduced Solvents

If substantial *in situ* biodegradation of highly chlorinated parent solvents (e.g., PCE and TCE) has occurred, and more-reduced (i.e., less-chlorinated) biodegradation daughter products (e.g., DCE and VC) dominate the groundwater plume, then creation of an aerobic treatment zone should be considered. This can be accomplished using proven technologies such as air sparging, bioventing, or injection of an oxygen source such as hydrogen peroxide or potassium permanganate. Addition of hydrogen or a hydrogen-releasing compound, as described in Section 4.2.3.1, also has been shown to be effective for degrading DCE and VC dissolved in groundwater.

Air Sparging. Air sparging involves injecting air under pressure into an aquifer with the objective of forcing the air to disperse through contaminated aquifer materials and groundwater (Figure 4.6). Treatment may occur either through volatilization or through aerobic biodegradation processes stimulated by adding oxygen (Downey *et al*., 1999). Although air sparging has been applied at numerous sites, the current understanding of air sparging performance and effectiveness is limited. One potential concern is the tendency for injected air to form channels in the aquifer, resulting in non-homogeneous treatment of the groundwater. Relatively homogeneous, sandy aquifers with shallow groundwater contamination may provide for more uniform treatment than sites with heterogenous mixtures of sand, silts, and clays. Air sparging systems typically are combined with SVE systems to collect vaporized contaminants that migrate into the vadose zone as a result of air injection below the water table.

Aerobic Biodegradation. Another relatively common bioaugmentation approach is the stimulation of direct aerobic oxidation of CAHs via biological or chemical mechanisms. This approach involves the addition of oxygen or an oxygen-releasing compound to the affected portion of the aquifer (see Section 4.2.3.1). As described in Section 2, aerobic enhancements are most useful for stimulating the aerobic biodegradation of DCE and VC in plumes where reductive dechlorination of these compounds is incomplete.

Air Sparging System



Bioventing System

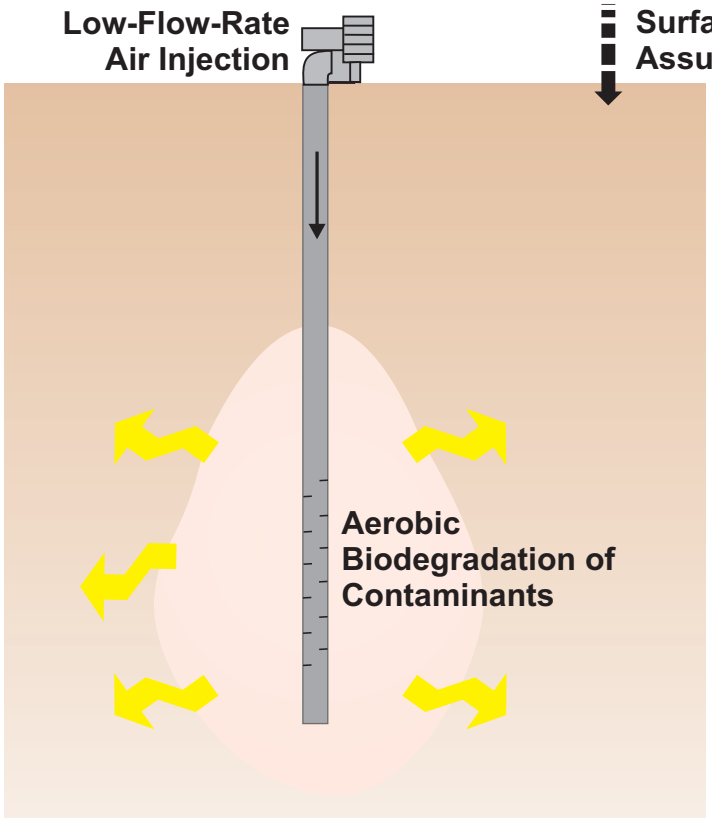


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FIGURE 4.6

TYPICAL AIR SPARGING AND BIOVENTING SYSTEMS

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Bioventing. Bioventing is the process of low-flow-rate (e.g., 5 to 20 cubic feet per minute) air injection into the vadose zone to supply oxygen for aerobic biodegradation of residual contaminants (Figure 4.6). However, the indiscriminate use of air injection at MSW landfills can increase the risk of spontaneous combustion within the landfill. A six-well air-injection pilot test conducted at a MSW landfill in Minnesota was effective in creating an aerobic biotreatment zone for landfill leachate as it migrates to groundwater (Barr *et al*., 1997). Results showed that organic compounds were actively biodegraded, and dissolved inorganics such as metals were oxidized and immobilized. However, bioventing probably is minimally effective at increasing the oxygenation of groundwater or for enhancing biodegradation of dissolved contaminants (Downey *et al*., 1999). Therefore, bioventing should be evaluated primarily as a source-removal technique.

4.2.3.3 Sequential Anaerobic/Aerobic Treatment Zones

Work by Turpie *et al*. (2000) on a contaminant plume at a Superfund landfill in New England highlighted the potential benefits of creating sequential anaerobic/aerobic treatment zones. Highly chlorinated solvents that are amenable to reductive dechlorination were transformed within the anaerobic zone (in this case via injection of sodium benzoate, which acts as an electron donor). In the event of incomplete transformation, less-chlorinated solvents migrating from the anaerobic zone will be aerobically degraded in the downgradient aerobic zone. According to the authors, implementation of this remedial approach will eliminate the need to extract, treat, and dispose of the contaminated groundwater. Sequential anaerobic and aerobic treatment zones can be created, as described in Sections 4.2.3.1 and 4.2.3.2, respectively (Figure 4.5).

4.2.4 Bioreactor Options

If the dissolved CAH plume can be contained by extracting contaminated groundwater that can be reapplied within the landfill, then leachate recirculation and operation of the landfill as a bioreactor should be considered. The various elements of a landfill bioreactor are conceptually depicted on Figure 4.7. In an unlined landfill, releases of potentially hazardous COCs can continue for decades. Available options for controlling these releases (e.g., low-permeability caps) do not reduce contamination at the source; therefore, these release controls may be required indefinitely, and at great expense for maintenance and monitoring.

Operation of the landfill as a bioreactor provides an *in situ* bioremediation alternative for problematic landfills that can potentially accelerate waste stabilization and contaminant-mass reduction processes at a relatively low cost. For older landfills, leachate recirculation may encourage further degradation of persistent chemicals by providing more favorable conditions for the microorganisms of interest (Harper and Just, 1995).

Consideration of the bioreactor option involves answering the following question (see Figure 4.2): "*Can an effective bioreactor be created within and/or beneath the landfill via recirculation of leachate-contaminated groundwater, with or without addition of amendments?*" Successful implementation of leachate recirculation depends on:

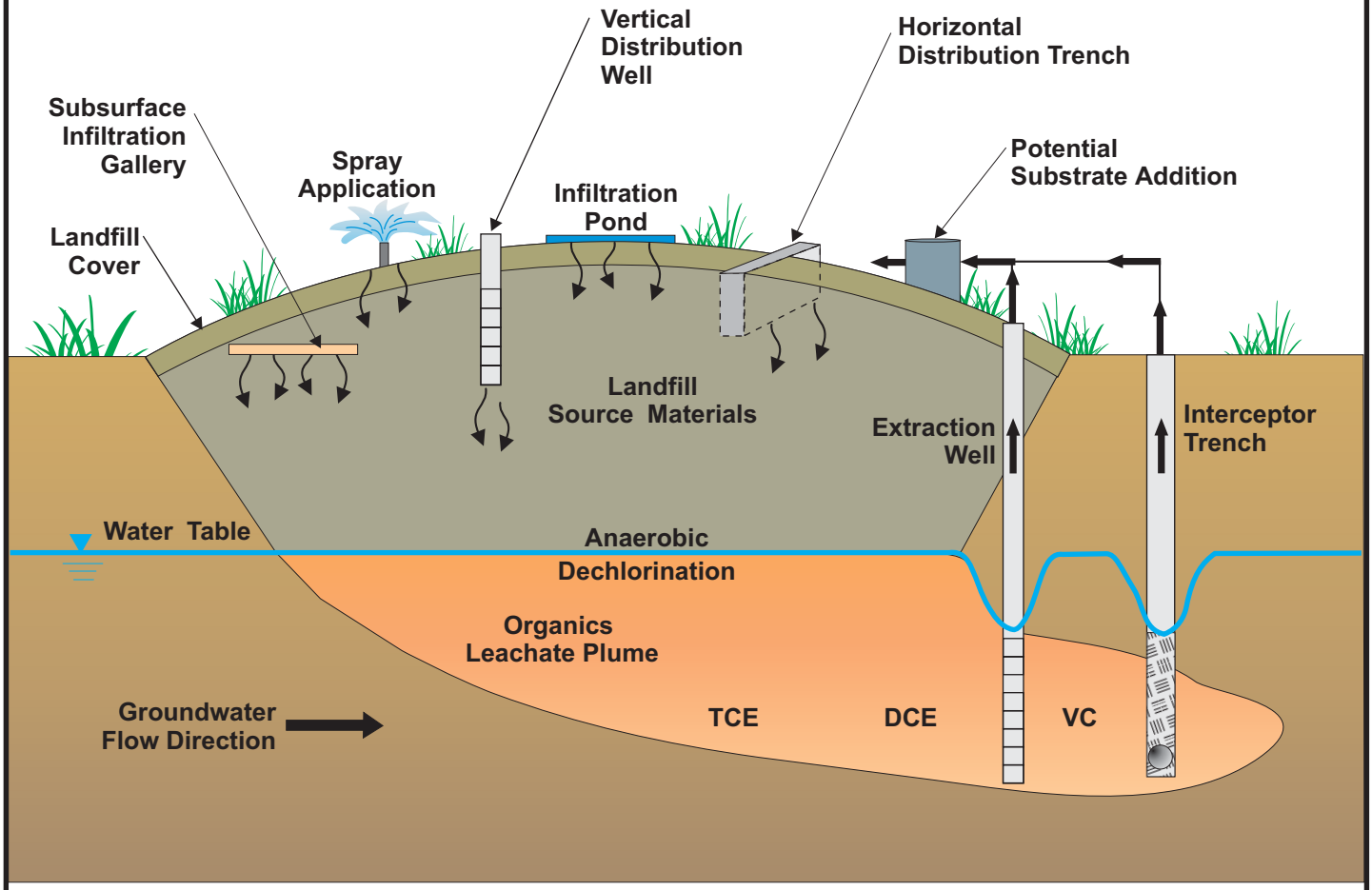

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FIGURE 4.7

SCHEMATIC CROSS-SECTION OF LANDFILL BIOREACTOR COMPONENTS

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- Collecting and controlling leachate as it is generated;
- Effectively redistributing leachate throughout the landfill without creating leachate seeps or other exposure-related issues;
- Controlling recirculation in a fashion that promotes robust microbial activity; and
- Containing, collecting, and managing/treating gases (e.g., methane) produced as a result of biodegradation.

These four criteria are discussed in Sections 4.2.4.3 through 4.2.4.6.

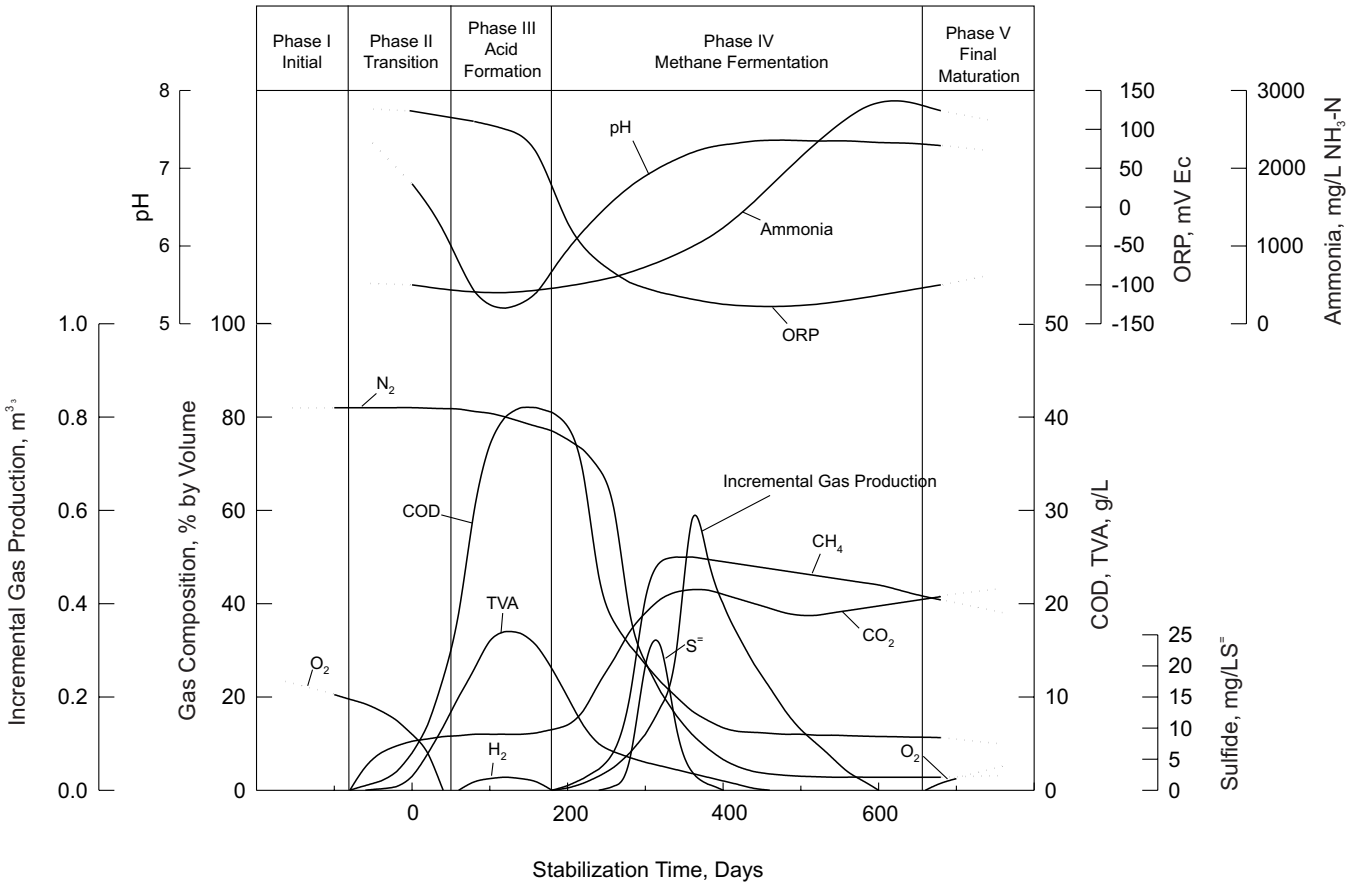
4.2.4.1 Landfill Stabilization Phases

Co-disposal of waste organic compounds such as CAHs with domestic and commercial refuse often results in their release during a series of five sequential stabilization phases. These phases are depicted on Figure 4.8 as reflected by concentration trends exhibited by common gas and leachate indicator parameters. As demonstrated by Pohland *et al*. (1993), associated transformation of co-disposed organic waste constituents, including CAHs, is superimposed upon this pattern of stabilization and results from conditions favorable for reductive dechlorination. Many DoD landfills are likely undergoing the final stages of Phase V waste stabilization (Final Maturation), which is characterized by a transition from active biological stabilization of readily degradable organics to relative dormancy and the possible transition from anaerobic to anoxic or aerobic conditions.

Minimum dissolved oxygen (DO) concentrations measured in groundwater beneath or immediately adjacent to six USAF landfills ranged from <0.1 mg/L to 2.1 mg/L and averaged 0.8 mg/L, indicating the presence of generally oxygen-depleted conditions ranging from anaerobic to slightly aerobic (Appendix B). Chemical oxygen demand (COD) concentrations measured downgradient from three of the landfills were relatively low (<5 to 28.9 mg/L [<0.005 to 0.029 grams per liter]), suggesting that these sites had progressed beyond the initial stages of methane fermentation (Phase IV, Figure 4.8).

As shown in Table 4.2, the mean groundwater pH value measured within or downgradient from six Air Force landfills was 7.2. This value is similar to the characteristic pH for the later stages of methane fermentation (Phase IV) or the initial stages of final maturation (Phase V) shown on Figure 4.8. In contrast, the mean groundwater ORP (-2 mV) was more oxidizing that the typical Phases IV and V values indicated on Figure 4.8. This is due in part to the high ORP values measured at FE Warren AFB LF03. If data from this site are excluded, the mean ORP measured within or downgradient from the remaining five landfills was –30 mV. Deviations from "typical" parameter values derived for municipal landfills are not surprising given the substantial diversity of waste types found at military landfills.

In the waste-stabilization process, prolonging Phase IV is typically of primary interest with respect to leachate recirculation because the associated geochemical conditions are most favorable for reductive dechlorination and cometabolism of CAHs. One of the keys



Ch_4 = Methane
CO_2 = Carbon Dioxide
COD = Chemical Oxygen Demand
g/L = Grams per Liter
H_2 = Hydrogen
M^3 = Cubic Meters
mg/L = Milligrams per Liter
mV = Millivolts
N_2 = Nitrogen
NH_3 = Ammonia
O_2 = Oxygen
ORP = Oxidation Reduction Potential
$S^=$ = Sulfide
TVA = Total Volatile Acids

Source: Pohland and Kim, 2001.

FIGURE 4.8

CHANGES IN SELECTED INDICATOR PARAMETERS DURING THE PHASES OF LANDFILL WASTE STABILIZATION

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to optimizing CAH degradation rates is to provide conditions conducive to methanogens, which have been shown to effectively dechlorinate CAHs. Addition of moisture is one of the primary means of accomplishing this objective.

TABLE 4.2
SUMMARY OF SELECTED GROUNDWATER GEOCHEMICAL RESULTS OBTAINED WITHIN OR DOWNGRADIENT FROM AIR FORCE LANDFILLS
MILITARY LANDFILL CLOSURE DESIGN EVALUATION

	COD[a/] (mg/L)[c/]	ORP[b/] (mV)[d/]	pH
Range	<5 to 28.9	-135 to 177	5 to 12.8
Mean	14.2	-2	7.2

[a/] COD = chemical oxygen demand.
[b/] ORP = oxidation-reduction potential.
[c/] mg/L = milligrams per liter.
[d/] mV = millivolts.

4.2.4.2 Determination of Primary Risk-Driver Chemicals

Application of the bioreactor scenario is dependent on the nature of the COCs in the plume. DCE can be biodegraded both aerobically and anaerobically, and it is not clear that one type of degradation process is typically more rapid than the other. Therefore, remedial decisions for DCE must be made on a site-specific basis using decay-rate information gained from the natural attenuation TS (Section 4.1.2). If DCE and/or VC are the primary COCs, the remedial goal should be to promote aerobic conditions to more efficiently degrade these compounds, as described in Section 4.2.3.2. This goal is not compatible with the bioreactor approach as it is typically applied, because recirculation generally promotes anaerobic, reducing conditions within the interior of the landfill due to the organic-carbon-rich nature of the recirculated leachate. Therefore, if PCE and/or TCE are the primary COCs, bioreactor operation is potentially advantageous.

However, bioreactor landfills can conceivably be managed to promote either anaerobic conditions (for PCE and TCE) or aerobic conditions (for DCE and VC). Aerobic conditions can be created or maintained by recirculation of naturally aerobic or mechanically aerated groundwater, or via addition of oxygen or an oxygen source into the subsurface. Methods for accomplishing these divergent objectives are discussed in Section 4.2.4.5.

4.2.4.3 Leachate Collection and Control

The major factor in deciding if CAH-contaminated groundwater can be reapplied within a landfill is the capacity of the landfill to accept water from the plume without adversely affecting the site. In order to make this decision, site-specific hydraulic models of landfill leachate production and contaminant transport within the affected groundwater system area should be developed to characterize the current conditions of the site. The models can then be used to determine the maximum amount of water that can be recirculated into the landfill without causing mounding and radial flow that could 1) alter the local groundwater flow and spread the plume in unwanted directions; 2) exceed the water-holding capacity of the landfilled wastes; and 3) potentially cause surface leachate seeps. Numerical models capable of simulating three-dimensional (3-D) transient flow within a heterogeneous, anisotropic environment should be used to prepare detailed designs of groundwater collection and distribution systems. However, simpler two-dimensional (2-D) analytical models that assume homogenous, isotropic conditions may, in some cases, be used to provide a preliminary simulation of these systems for conceptual-design purposes.

The Hydrologic Evaluation of Landfill Performance (HELP) model (Schroeder *et al*., 1994) was developed specifically for water-balance analysis of landfills, and can be used to facilitate the estimation of the amounts of runoff, ET, and leachate formation that may be expected to result from the operation of a wide variety of landfill designs (Wong *et al*., 1997). However, Suthersan (2001) reports that the HELP model was developed based on assumptions pertaining to water management through low-permeability soil covers with short-rooted grass vegetation. Therefore, if plant species with significantly higher ET rates are used, as would be the case for an ET cover (see Section 4.3), then use of the HELP model will result in an overestimation of infiltration rates. Other potentially useful models, including one developed to evaluate the performance of an engineered ET cover, are described by Suthersan (2001) and USEPA (1999a).

Once the capacity of the landfill to receive water is established, the contaminant transport model would be used to determine the groundwater extraction rate necessary to capture the CAH plume, and to design the extraction system (i.e., number and location of extraction wells or trenches, and appropriate pumping rates). If the capacity of the landfill to receive water exceeds the flow necessary to capture the plume, or if recirculation of a portion of the contaminated groundwater is feasible, then recirculation merits further evaluation. The numerical groundwater flow model MODFLOW (McDonald and Harbaugh, 1988), coupled with the contaminant transport model MT3D (S.S. Papadopulous & Associates, 1996), are perhaps the models most commonly used to simulate groundwater flow systems and contaminant plume fate and transport. WINFLOW (Environmental Simulations, Inc., 2000) is an example of a 2-D analytical model used to simulate groundwater flow and contaminant transport.

The optimal method for plume capture is dependent on the characteristics of the contaminants and their distribution. If dissolved CAH contaminants are the result of a DNAPL source, the DNAPL and resulting dissolved plume generally extend to the top of a confining layer (bottom of the aquifer). In such cases, capture of the CAHs usually requires a fully penetrating well screen and a specialized pumping system that allows for

recovery of both DNAPL and groundwater. Sumps may be required within extraction wells set into the confining zone (Fetter, 1999).

Sites without significant DNAPL or with slowly releasing DNAPL may not affect the full depth of the aquifer. At such sites, dissolved CAH contamination can be limited to the upper portion of the aquifer or may be transported in shallow transmissive zones.

In landfills, the DNAPL source usually is contained within the landfill, and the plume is generally composed of dissolved CAHs along with other contaminants. The optimum plume-capture techniques for this situation are determined by the location of the plume within the aquifer, as well as the chemical and physical properties of the predominant COCs within the plume.

To date, conventional pump-and-treat approaches to DNAPL removal and source zone restoration have proven to be largely ineffective (Pankow and Cherry, 1996). The main benefit that pump-and-treat offers at DNAPL sites is plume or source zone containment. As a result, chemical treatment processes using surfactants or co-solvents have been used to increase the effective solubility of DNAPL so that circulation of a water/chemical mixture through the DNAPL zone will remove contaminant mass more rapidly than conventional pump-and-treat (Pankow and Cherry, 1996). Under certain conditions, surfactants or co-solvents also can be used to reduce the interfacial tension that traps DNAPL in porous media, resulting in mobilization of the DNAPL and an increased potential for source mass reduction either through pump-and-treat or enhanced biological treatment (Hughs *et al*., 2001). The potential exists for residual co-solvent to stimulate *in situ* bioremediation of CAHs following partial DNAPL source removal (Mravik, 2000).

Groundwater Extraction Trenches. For shallow plumes, extraction trenches can be more cost-effective than extraction wells, particularly in strata with low or variable hydraulic conductivity. Under these conditions, it would be difficult to design and cost-prohibitive to operate an extraction-well system that would maintain continuous hydraulic containment of the plume. Trenches may also be preferred over wells where groundwater removal over a period of several years is required, because the O&M costs associated with wells are substantially higher than those for trenches. One of the biggest drawbacks to the use of extraction trenches is their depth limitations. Trenches may be cost-prohibitive at depths of greater than 40 feet below ground surface (bgs); however, in stable, low-permeability formations where no rock excavation, shoring, or dewatering is required, trenches may be cost-effective to depths of 100 feet (USEPA, 1985). If aeration of the collected groundwater in the trench is to be avoided, the trench should be designed to minimize contact of groundwater with the atmosphere.

Groundwater Extraction Wells. Well systems are very versatile and can be used to hydraulically contain and remove plumes under a wide variety of site conditions. However, the geometry and extent of pumping-well capture zones is influenced by aquifer hydraulic conductivity and heterogeneity; therefore, extraction wells are best suited for relatively conductive and homogeneous aquifers where laterally extensive and uniform capture zones can be developed. Wells should be operated to minimize aeration of the extracted water to the extent feasible. For example, centrifugal pumps can cause excess aeration, while diaphragm pumps cause less aeration. Low-energy pumps are

preferred for capturing plumes consisting of highly-chlorinated solvents such as TCE and PCE due to the need to keep the water as anaerobic as possible for recirculation purposes.

Aquifer heterogeneities can include strata with variable hydraulic conductivities, such as silt or clay lenses in sandy strata, which can cause preferential flow paths. Water extracted from the aquifer would be drawn into the wells primarily from these preferential flow areas, and extraction of any contaminants partitioned in the lower-conductivity zones would take much longer. Such heterogeneities may require the extraction system to include wells constructed specifically to capture water from these low-conductivity zones. However, partial capture of groundwater from the most contaminated (and permeable) strata may be acceptable at sites where MNA is capable of remediating lower levels of CAHs.

Influence of Plume Depth on Capture Efficiency. The location of the plume is important in the design of any capture system. Capture-systems designed for shallow plumes (at or near the water table) generally are much easier to construct and operate, and provide a wider range of extraction options (e.g., wells, trenches, or well points) than do deeper plumes. Deeper plumes generally require the construction of wells, which can be expensive. Deeper plumes also have a greater chance of affected heterogeneous aquifer materials, which may require more wells screened at multiple depths in order to remove the plume. Depending on the depth of the contamination, the extraction system may also require the use of relatively expensive submersible pumps because suction pumps have a finite depth to which they are efficient. Deeper plumes may not be able to be economically removed, especially if the hydraulic conductivity of the affected aquifer is sufficiently low that wells would require an excessively long time to recharge.

Capture-Zone Estimation. Determination of the best well-pattern and capture-zone design for plume cleanup or capture is highly site-specific (Fetter 1999; Satkin and Bedient, 1988). Therefore, it is difficult to provide specific guidance on the design of capture zones for landfill CAH-plume applications. The most common analytical model used for capture-zone analysis is that of Javandel and Tsang (1986), which assumes a homogeneous medium. However, significant error can be involved in applying the homogeneous assumption to heterogeneous media (e.g., Fetter, 1999; Guadagnini and Franzetti, 1999; Satkin and Bedient, 1988). As demonstrated by Shafer (1987), even relatively simple heterogeneities can result in significant asymmetry and irregularity of a capture zone compared to those predicted for homogeneous media. Examples of the use of numerical techniques for capture-zone analysis are provided by Satkin and Bedient (1988), Shafer (1987), and Guadagnini and Franzetti, (1999). It is recommended that 3-D numerical groundwater-flow and contaminant-transport models be used to assess the effectiveness of capture-zone and recirculation applications at DoD MSW landfills for final, detailed design purposes. However, simpler analytical models sometimes can be used to obtain approximations for use during the conceptual design phase. Once capture-zones designs are implemented, frequent monitoring should be implemented to assess the actual performance of the design.

4.2.4.4 Leachate Application to the Landfill

Evaluation of leachate-recirculation must include assessment of where, how, and at what rate leachate should be introduced into the landfill. These issues are discussed in the following paragraphs.

Leachate Application Areas. Leachate recirculation need not be performed in areas that did not receive biodegradable waste (e.g., that received construction debris), or where groundwater and soil-gas chemistry indicate that the waste-stabilization (degradation) process is essentially complete (e.g., low methane concentrations, see Figure 4.8). Conversely, recirculation should be focused in areas that contain high levels of CAHs, and high biological oxygen demand (BOD) and COD; low-moisture content that may be inhibiting bioremediation; or any other characteristic that may be limiting biodegradation (e.g., aerobic conditions in areas where anaerobic conditions are desired).

Leachate Application Methods. Potential leachate-application techniques include vertical wells and horizontal infiltration trenches for anaerobic applications and infiltration ponds or leach fields and spray-application onto the landfill surface for aerobic applications (Figure 4.7). All of these methods have been considered and/or applied at various landfill sites in the past.

The easiest and least costly method of aerobic leachate recirculation would be by surface irrigation, which has an added potential benefits for irrigating cover vegetation. However, where emission of volatile compounds (e.g., VC) into the atmosphere is a concern, a subsurface injection method (e.g., injection wells and/or trenches, using conventional design and installation techniques) may be preferable.

Harper and Just (1995) note that vertical infiltrations wells (perforated pipe backfilled with gravel, scrap tires, or other materials) facilitate infiltration of leachate through daily soil-cover layers and layers of other low-permeability material. A lateral dispersion of 1 foot per 4 feet of vertical percolation was assumed for design purposes. In contrast, Miller and Emge (1997) describe a situation where a horizontal trench system was preferred over a well system because it was the easiest to implement, and installation could be coordinated with that of the landfill-gas collection system, and with landfill operations. Horizontal systems may be preferred at new landfills where the leachate recirculation system can be incorporated into the landfill as it is constructed, or at active landfills as they are filled. In these cases, leachate-distribution piping can be laid at multiple levels within the landfill as it fills, allowing proper vertical distribution of recirculated leachate within the landfill. Miller and Emge (1997) also state that aerated leachate may also be recirculated to the landfill by surface spray application. This is expected to provide substantial leachate volume reduction by ET. The authors note that aeration should not be used in combination with subsurface leachate recirculation, but should be provided before surface application to reduce odors. An existing lined landfill in Florida was converted to a bioreactor treatment system via construction of four leachate infiltration ponds excavated into the landfilled waste (Townsend *et al*., 1996).

Full-scale application of leachate recirculation has been hampered by an inability to uniformly apply leachate to the waste mass, due in large part to the heterogeneous nature of the disposed waste, which promotes preferential routing and short-circuiting of

leachate flows (Johnson *et al*., 1998, McCreanor and Reinhart, 2000). Younger landfills are more likely to exhibit preferential flow. As the landfill ages, the refuse biodegrades and settles, the medium gets more homogeneous, reducing the impact of fast, preferential flow (Bendz *et al*, 1997).

The observations recorded in the literature suggest the following guidelines for leachate recirculation into typical DoD MSW landfills:

- The nature of the subsurface should be characterized to the extent possible via review of historical waste-disposal records, soil borings, and/or test pits to determine the presence/absence of low-permeability layers that would impede vertical migration of recirculated water, and would make vertical wells the recirculation method of choice.

- Surface application of CAH-contaminated groundwater via spraying or infiltration ponds may be undesirable at many sites due to the potential for odor concerns; freezing (in some geographic areas); adverse exposure via surface runoff, volatilization, or dermal contact; and aeration of the water (undesirable for highly chlorinated CAHs such as PCE and TCE; potentially desirable for DCE and VC). In addition, there is a potential for formation of a low-permeability layer at the ground surface or the bottom of infiltration ponds due to oxidation and precipitation of dissolved constituents in the leachate-contaminated water (e.g., iron) that can inhibit infiltration.

- Horizontal trenches, vertical wells, or buried leach fields may be the most suitable alternatives at most DoD landfills. A combination of shallow horizontal and deeper vertical delivery mechanisms (e.g., trenches and wells) would overcome the limitations of each individual approach. Maintenance of anaerobic conditions will promote reductive dechlorination of CAHs and reduce fouling from oxidized metals such as iron.

- It may be desirable to implement recirculation in a phased approach, with initial application in a test plot to evaluate the impacts on the degradation of organics and to test leachate application rates.

Leachate Application Rates. Techniques to determine the optimal rate of leachate application are not well documented in the literature. If recirculation rates are too high, the landfill could become flooded with leachate over time. The ability of the landfill to absorb recirculated leachate without requiring offsite disposal influences the cost/benefit analysis.

The following steps are recommended by and described in Harper and Just (1995) to estimate leachate recirculation rates:

1. Determine leachate-generation rates;

2. Select an allowable head build-up within the landfill;

3. Evaluate prevailing landfill moisture conditions; and

4. Estimate the period of time to reach storage capacity.

A more detailed evaluation of a landfill's response to recirculation may be attained by modeling, as described in Section 4.2.4.3, although accurate modeling of the interior of a poorly characterized, heterogeneous landfill will likely be problematic. Due to the difficulties inherent in predicting the response of a landfill to recirculation, Harper and Just (1995) recommend designing the recirculation system to be flexible and adjustable, and to monitor the effects of recirculation over time.

4.2.4.5 Enhancement of Microbial Activity and Construction of Dedicated Treatment Zones

Microbial activity within landfills can be promoted by the addition of moisture and specialized microbial populations, and by the engineered creation/enhancement of aerobic or anaerobic conditions that are conducive to the occurrence of desirable chemical reactions. Engineered adjustment of pH conditions and nutrient levels also can foster conditions conducive to microbial activity. Site conditions and the type of biotransformation process to be advanced will determine if the use of amendments would be appropriate at a given landfill, and which amendments should be applied.

The following steps typically are used in the selection and implementation of *in situ* bioremediation (USEPA, 2000e):

- Evaluate site characteristics;
- Identify general site conditions and engineering solutions;
- Identify primary reactants and possible additives;
- Perform treatability testing; and
- Perform system design, field testing, and implementation.

A dedicated treatment zone within the landfill could be constructed, and leachate-contaminated groundwater could be directed toward this zone for treatment. Such a system could be operated in a batch-type mode until the target COCs are transformed. The treated effluent from this zone could serve as a transport medium to accelerate leaching of residual waste constituents as it is recycled through the landfill matrix. Flexibility in the design, location, and operation of the dedicated treatment zone could accommodate a variety of treatment scenarios for removal of CAHs from both aqueous leachate and vapor phases.

Because the leachate collection and recirculation system is the principal operational technique needed to create and operate a treatment zone, the characteristics of the landfill leachate (and gas) can direct the choice of transformation mechanism to be promoted, and influence the need for augmentation. For example, if leachate quality indicates that stabilization of readily degradable organic substrates is nearing completion (i.e., Phase V, Figure 4.8), promotion of anaerobic cometabolic dechlorination under nitrate- and sulfate-reducing conditions may be indicated. If deficient in the leachate, electron

acceptors could be added prior to recirculation, as has been successfully demonstrated for soil and groundwater remediation (Table 4.3). Replacement of anaerobic or anoxic conditions with aerobic conditions via aeration could also enhance methanotrophic cometabolic transformation of CAHs, or direct dehalogenation of DCE and VC. The use and location of engineered treatment zones would be a function of applicability and existing landfill construction and retrofitting opportunities.

Treatment zones at DoD MSW landfill sites can be engineered within and/or beneath the waste mass. For example, if groundwater beneath the landfill is anaerobic and contains a substantial concentration of dissolved organic carbon (DOC), then anaerobic conditions both within and beneath the landfill can conceivably be maintained or enhanced by collection and recirculation of unamended, leachate-contaminated groundwater through the waste mass. If amendments are required to enhance anaerobic conditions, they can be added to the recirculated leachate and/or injected directly into the groundwater beneath or downgradient from the landfill (i.e., within the CAH plume area). The advantage of amending the recirculated water is that this would promote biodegradation of residual contaminants within the unsaturated portion of the landfill, and would result in percolation of anaerobic, reducing groundwater into the saturated zone. This approach may also be more cost-effective than direct injection of amendments into deeper aquifer zones, and would allow for more controlled and complete mixing of the amendment with the water. However, if more aggressive, short-term treatment of a groundwater plume is required, then injection of amendments directly into the groundwater beneath the landfill should be performed. A review of the literature (Section 3) indicates that direct injection of amendments into the groundwater is the most common approach implemented to date.

Moisture Addition. Moisture content is probably the most important factor governing the biochemical processes in a landfill. According to Harper and Just (1995), the most effective strategy for stimulating methanogenic activity (including reactions that result in the biotransformation of CAHs) involves moisture addition. In cases where less-than-optimal moisture content in the landfilled wastes is limiting biotransformation of CAHs, moisture can be added via recirculation of leachate-contaminated groundwater. "Dry" landfills, where moisture is excluded or where field capacity is not exceeded, could store the waste mass, including any CAHs, for an indefinite period, resulting in a prolonged and uncertain release prognosis, and environmental impact. This uncertainty could be reduced by converting the landfill to a more biochemically active state, perhaps with leachate collection and recirculation.

Natural attenuation involves a combination of processes that can be shown to occur or be selected for in landfill disposal systems. Whereas reductive dehalogenation, oxidative aerobic biotransformation, cometabolic transformation, sorption, volatilization, dispersion, and abiotic processes act upon dissolved plume contaminants in natural and engineered remediation systems, similar reaction opportunities generally exist at the landfill waste source, particularly if moisture is sufficient to support microbially mediated transformations and transport of CAHs and their transformation products in either the gas or liquid phase. When leachate containment, collection, and recirculation are employed, as in the case of landfill bioreactors, reaction opportunities can be manipulated to create more rapid and thorough waste stabilization and CAH treatment at the source (Pohland and Kim, 2000). Addition of water may increase the bioavailability of organic

TABLE 4.3
COMPONENTS OF *IN SITU* BIOREMEDIATION TECHNOLOGY
MILITARY LANDFILL CLOSURE DESIGN EVALUATION

Component	Example	Biodegradation Mechanisms Supported	Target CAHs [a/]
Bioaugmentation	Seed subsurface with non-native CAH-degrading bacteria.	Aerobic oxidation (cometabolic and direct)	TCE, DCE, VC, TCA, DCA, CA, CT, CF
		Anaerobic reductive dechlorination (cometabolic and direct)	TCA, DCA, CA, CT, CF, CM
Addition of Nutrients	Add nitrogen, phosphorus, or other growth factors that may be deficient in the subsurface	Aerobic oxidation (cometabolic and direct)	TCE, DCE, VC, TCA, DCA, CA, CT,CF
		Anaerobic reductive dechlorination (cometabolic and direct)	TCA, DCA, CA, CT, CF, CM
Addition of Electron Donors	Add a substrate, such as toluene, propane, or methane	Aerobic oxidation (cometabolic)	TCE, DCE, VC, TCA, CF, CM
	Add hydrogen, a hydrogen source, or a hydrogen-releasing compound	Anaerobic reductive dechlorination (cometabolic and direct)	PCE, TCE, DCE, VC, TCA, DCA, CA, CT, CF, MC
Addition of Electron Acceptors	Add oxygen by bioventing, biosparging, or adding an oxygen source such as hydrogen peroxide	Aerobic oxidation (direct)	TCE, DCE, VC, TCA, DCA, CA, CE?, MC, CM, VC
	Add an anaerobic reductant such as nitrate	Anaerobic reductive dechlorination (cometabolic)	PCE, TCE, DCE, VC, DCA, CT

Sources: Sewell *et al*., 1998; USAF, 1998; USEPA, 2000e.

[a/] PCE = tetrachloroethene, TCE = trichloroethene; DCE = dichloroethene; VC = vinyl chloride; TCA = trichloroethane; DCA = dichloroethane; CA = chloroethane; CE = chloroethane; CT = carbon tetrachloride; MC = methylene chloride; CF = chloroform.

compounds, and remediation methods designed to reduce moisture infiltration could adversely affect and prolong contaminant biodegradation at the source (Sanin *et al*., 2000).

In some landfills, CAHs are undergoing transformation, as indicated by the presence of PCE/TCE as well as DCE/VC, but the waste saturation is below field capacity and there is no evidence of leaching. Such landfills could benefit from judicious management of infiltration/ET in order to accelerate dehalogenation processes without leachate production and possible plume formation. In such cases, it would be feasible to increase the degree of waste saturation by allowing infiltration (of natural precipitation and/or irrigation water) to slightly exceed ET and runoff. This could be accomplished by installing a permeable moisture-receptor cover system that would promote infiltration while serving as a biofilter for treatment of potential gas and volatile CAH (e.g., VC) emissions. When vapor monitoring indicates that CAH sources have been removed, the permeable cover could be replaced by a lower-permeability cover to limit further infiltration and allow ET and runoff to control moisture. Thereafter, the closure strategy would be similar to that prescribed for a relatively stable landfill.

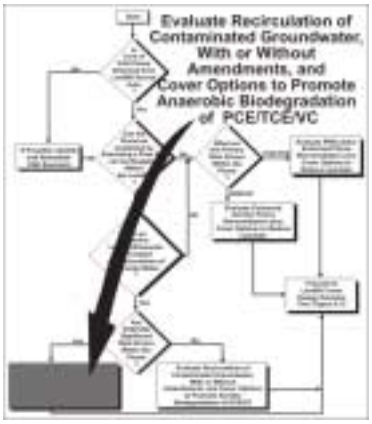
To prevent potential leachate production in excess of field capacity, the moisture-loading schedule would need to be managed to allow accommodation of contaminated water/leachate at quantities equal to the amount removed and/or handled by the site. Accordingly, the amount of liquid introduced to stimulate *in situ* treatment could again be equivalent to any field-capacity deficit plus that consumed by potential ET and biological utilization. Hence, the selected design would depend upon the landfill setting and associated site-specific factors, including climate, hydrogeology, waste matrix, and contaminant source characterization.

Contaminant source delineation and possible removal (to the extent possible) will also influence the selected operational protocol for a leachate-recirculation system. Clearly, uncontrolled acceleration of contaminant release and possible plume extension must be avoided. Moreover, the frequency of leachate loading also would depend on the rate of groundwater extraction required to contain the plume, and could involve either continuous or intermittent operations. If a dedicated landfill treatment zone is used, it would likely be sized to be operated in a fill-and-draw fashion (i.e., adding moisture until field capacity is reached, then halting moisture addition to permit sufficient residence time for the current moisture load to promote the desired microbial processes before further moisture addition is performed) on a schedule that permits the desired transformation efficiency for the volume of waste (mass of CAHs) to be treated. Therefore, the capacity of such a dedicated zone would be a function of desired hydraulic residence time and effluent-reception constraints.

Addition of Microbes (Bioaugmentation). Based on the documented ability of naturally occurring microorganisms to degrade chronic contamination (especially at older waste-release sites such as DoD landfills, where the native microbial populations have become fully acclimated to the contaminants), the addition of specialized bacteria to enhance biodegradation processes is not recommended. With few exceptions, natural bacteria are capable of degrading contaminants at the same (or faster) rates as foreign, introduced bacteria (Downey *et al*., 1999). Suthersan (2001) reports that the record of success of *in situ* bioaugmentation systems for chlorinated compounds has been "spotty,"

and that the success rate has been much greater in laboratory bioreactors, where conditions may be much different than those encountered in nature.

Enhancement of Anaerobic Reductive Dechlorination. Direct and cometabolic reductive dechlorination of CAHs in anaerobic environments are discussed in Sections 2.4 and 2.6, respectively. It should be noted that cometabolic processes are typically less efficient than direct anaerobic dechlorination if highly chlorinated CAHs such as PCE and TCE are the COCs.



As described in Section 4.2.4.1, at older landfills, such as are present at most DoD installations, where significant oxidation of available organic carbon may have already occurred, anoxic or aerobic conditions may be present at least locally, although oxygen concentrations are often below background levels. The occurrence of below-background DO concentrations in groundwater was documented at six typical USAF landfills (see Section 4.2.4.1). The average minimum DO concentration measured at these sites was 0.8 mg/L, indicating the effects of oxygen-depleted leachate.

Creation/enhancement of anaerobic conditions within and/or beneath the landfill may be desirable for two reasons:

1. Highly chlorinated solvents such as PCE and TCE, which are primary COCs in many DoD landfills, can be efficiently dechlorinated in anaerobic environments, either directly or via cometabolism. Available information indicates that direct dechlorination is probably more rapid and efficient than cometabolic dechlorination, but distinguishing between the two processes is difficult at the field scale. TCE (though not PCE) also can be cometabolically transformed in aerobic environments, but this process may be less efficient and rapid than direct anaerobic reductive dechlorination.

2. It makes sense to enhance biotransformation reactions that are already operative and significant in the landfill environment, and anaerobic reductive dechlorination is often a primary transformation mechanism in the oxygen-depleted interiors of many landfills. There generally is an adequate supply of fermentation substrates for the production of dissolved hydrogen, which is a prerequisite in order for halorespiration to play a significant role in attenuating CAH compounds.

Creation/enhancement of anaerobic conditions via electron-donor addition involves the addition of a substrate that acts as a reductant in the redox reaction by using the CAH-degrading microbe to produce energy. A substrate such as toluene, propane, or methane may be added to act as a cometabolic oxidant, when the CAH also is oxidized. A substrate such as hydrogen, a source of hydrogen, or HRC® may be added to act as a direct reductant, when the CAH is reduced (USEPA, 2000e). According to the USEPA (2000e), typical additives for enhancing anaerobic reductive dechlorination include lactate, methanol, hydrogen, and molasses. Injection of food-grade vegetable oil into the

groundwater also has been implemented at several sites in the US, and this promising approach is being actively tested by AFCEE.

Pilot studies at a variety of sites have successfully demonstrated the feasibility of stimulating *in situ* anaerobic reductive dechlorination by providing simple organic substrates, such as lactate, butyrate, methanol, ethanol, and benzoate to the subsurface (Harkness *et al*., 1999; Freedman and Gossett, 1989; Gibson and Sewell, 1992; Buchanan *et al*., 1997; Becvar *et al*., 1997; Sewell, 1998; Litherland and Anderson, 1997; Spuij *et al*., 1997).

Chlorinated solvents emanating from a landfill at a Superfund site in New England are being treated in a field-scale pilot study with a biostimulation approach that includes sequential anaerobic and aerobic treatment zones. Electron-donor (sodium benzoate) injection in the anaerobic zone at the toe of the landfill is designed to enhance the biodegradation of CAHs (Turpie *et al*., 2000). This approach has shown promise, and it allows precipitation to continue to infiltrate and percolate through the landfill, maintaining conditions that support the existing microbial populations.

Enhancement of Direct Anaerobic Oxidation. As stated in Section 2.3, a recent study has demonstrated significant anaerobic oxidation of VC to carbon dioxide under Fe(III)-reducing conditions (Bradley and Chapelle, 1998). The results suggest that addition of chelated Fe(III) to the groundwater system may be an effective method to enhance bioremediation of VC plumes. This bioremediation pathway for VC was only recently discovered, and other case studies describing the enhancement of this pathway via chelated Fe(III) addition were not encountered during the literature review performed for this project. In anaerobic environments where VC is accumulating, enhanced anaerobic oxidation can be implemented to degrade this compound via the addition of chelated Fe(III) to recirculated groundwater or injection directly into the affected aquifer.

Enhancement of Aerobic Cometabolic Oxidation. Cometabolic oxidation of CAHs is discussed in Section 2.5. Enhancement of this process would involve introduction of both oxygen and a carbon source into the subsurface. According to USEPA (2000e), additives typically used to create aerobic conditions include ambient air, oxygen, hydrogen peroxide, and magnesium oxide. Typical carbon additives include methane, propane, butane, and ammonia (organic carbon). Collectively, this oxidative mechanism could be operative in both the soil and landfill matrix provided that a sufficient hydrocarbon source is present or can be delivered to the indigenous microflora. This may be an objective in a modified landfill bioreactor system where both anaerobic and aerobic stabilization zones can be established, and leachate collection and recirculation is employed. Apart from explosion hazards, it would not be technically difficult to add one or more of these substrates to recirculated water. Aerobic cometabolic oxidation case studies involving the addition of various amendments are summarized in Section 2.5.

Enhancement of Direct Biological or Chemical Oxidation. Enhancement of direct aerobic oxidation of CAHs should be evaluated when less-chlorinated compounds (e.g., DCE and VC) are the COCs. Generally, direct oxidation mechanisms degrade CAHs more rapidly than cometabolic mechanisms (McCarty and Semprini, 1994). According to USEPA (2000e), typical additives for promotion/enhancement of direct aerobic oxidation include air or an oxygen source such as hydrogen peroxide. More than a single application of oxidant generally is required to meet most cleanup standards. Recently, continuous injection using recirculation of amended waters has been used to maximize the utilization efficiency of the oxidant and augment the distribution rate within the reactive zone (Suthersan, 2001).

Direct chemical oxidation involves the injection of strong oxidants such as hydrogen peroxide, Fenton's Reagent (hydrogen peroxide plus an iron catalyst), potassium permanganate, and ozone to contaminated soil, sludge, and groundwater. Target chemicals have included CAH compounds, semi-volatile compounds, polychlorinated biphenyls (PCBs), and pesticides (USEPA, 1998b). Because landfill leachates can exhibit significant non-hazardous COD, much of the injected oxidant could react with non-CAH compounds. Application of direct chemical oxidation to large plumes or landfill perimeters is much more expensive than biological enhancement techniques, particularly when plume treatment is extended over decades.

As mentioned in Section 4.2.3.2, Barr *et al*. (1997) describe a 6-month-long, six-well air-injection pilot test conducted to create an aerobic biotreatment zone for leachate migrating to groundwater. Test results showed that an effective aerobic treatment zone can be established and maintained with organic compounds actively biodegraded, and dissolved inorganics such as metals oxidized and immobilized. In addition, the authors noted that further biotreatment of leachate reaching the groundwater can continue by enhanced diffusion of oxygen from the vadose zone into the groundwater. However, such diffusion is generally minimal and relatively insignificant. It should be noted that the indiscriminate injection of oxygen into a landfill environment can increase the risk of spontaneous combustion within the landfill; therefore, injection must be performed under carefully controlled and closely monitored conditions.

Anaerobic Reductive Dechlorination Combined with Aerobic Oxidation. Mixed anaerobic and aerobic environments may coexist within a given landfill, depending on intrinsic waste and hydrogeologic heterogeneities, the influence of flow regimes, and the presence or absence of sources of contaminants, including CAHs, possibly mixed with landfill leachate constituents. Landfill remediation case studies summarized in Section 3 indicate that stratified redox conditions often develop to some degree beneath and downgradient from landfills. Such conditions could be promoted or enhanced, as appropriate, via recirculation of leachate-contaminated groundwater with concurrent addition of amendments (as described in previous subsections) to achieve the desired subsurface geochemical conditions. These mixed anaerobic/aerobic conditions represent the most likely scenario for successful biodegradation of highly chlorinated solvents such as PCE and TCE.

An idealized representation of redox zones ranging from anaerobic (methanogenic to nitrate- and manganese-reducing zones) to aerobic that may naturally develop in the saturated zone beneath and downgradient from a landfill is depicted on Figure 4.9. The

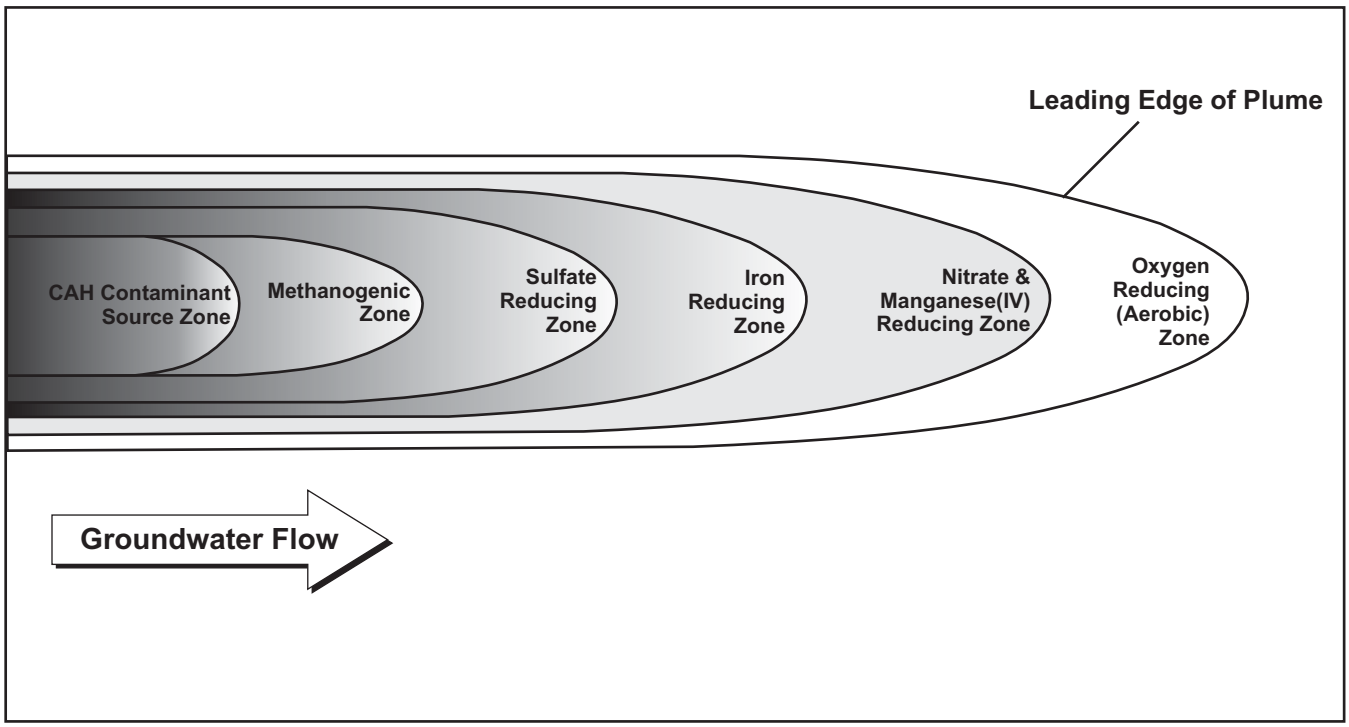


FIGURE 4.9

REDOX ZONES WITHIN CAH PLUMES

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relative proximity of these environments determines the stage and prominence of particular CAH transformation processes. These processes tend to complement each other, as can be observed by CAH transformation and the presence of associated transformation products. As a consequence, the gradation from anaerobic and aerobic mechanisms within close spatial relationships could allow direct and cometabolic mechanisms to be operative, leading to the total transformation of CAHs.

The interaction between the different anaerobic and aerobic transformation mechanisms was originally conceived for a continuum of aquifer conditions, including an interface between the aerobic and anaerobic zones at which, in the presence of both methane and oxygen, cometabolic oxidations convert the chlorinated ethenes to carbon dioxide and chloride. In contrast, in the aerobic zone, direct oxidation of VC and perhaps DCE could occur with oxygen or ferric iron, while in the anaerobic zone where organic electron donors and hydrogen are present, the predominant mechanism would be reductive dehalogenation yielding ethene and chloride.

Several investigators have suggested that the most efficient biotransformation of CAHs will occur in aquifers that are characterized by an upgradient anaerobic zone and a downgradient aerobic zone (Fathepure *et al*., 1987; Carter and Jewell, 1993; Bouwer, 1994; Gerritse *et al*., 1995;). In the upgradient zone, anaerobic reductive dechlorination of PCE might yield TCE, DCE, and eventually VC. DCE and VC could then be transformed by (direct) aerobic oxidation downgradient in the aerobic zone (i.e., the leading-edge or fringe) of the CAH plume. Such stratified redox conditions in the field may best describe the overall nature of intrinsic transformation of CAHs. A similar scenario could be conceived within modified landfill systems by incorporating enhanced anaerobic/anoxic and aerobic zones as shown on Figure 4.10.

The field-scale pilot study performed by Turpie *et al*. (2000) and described under "Enhancement of Anaerobic Reductive Dechlorination" (above) is being conducted using a biostimulation approach that includes sequential anaerobic and aerobic treatment zones. Electron donor (sodium benzoate) injection in the anaerobic zone at the toe of the landfill is designed to enhance the biodegradation of CAHs. Oxygen injection is used to create an aerobic treatment zone to enhance biodegradation of petroleum hydrocarbons and non-chlorinated solvents and to stimulate the precipitation of arsenic. This treatment system provides many advantages over remedial measures currently proposed for this and other similar sites with an impermeable cap and pump-and-treat leachate collection. Preliminary results indicate the dual-treatment-zone approach provides a mechanism for treating compounds that are best degraded under anaerobic as well as those degrade under aerobic conditions.

pH Adjustment. Leachate recirculation typically produces a residue potentially high in salts, heavy metals, and some organic compounds, and can inhibit methanogenic bacteria (and thus methane production) by contributing to the acidic conditions in the landfill if natural or added buffer (such as lime) is not present. As shown on Figure 4.8, the acid-formation phase of landfills (Phase III) is characterized by pH in the range of 5 to 6, whereas the methane-fermentation Phase IV is characterized by an increase of pH to neutral. USEPA (1998b) states that microbes capable of degrading CAHs generally prefer pH values varying from 6 to 8 standard units. Harper and Just (1995) state that methanogens favor neutral to slightly alkaline pH in the range of 7.0 to 7.8 standard units.

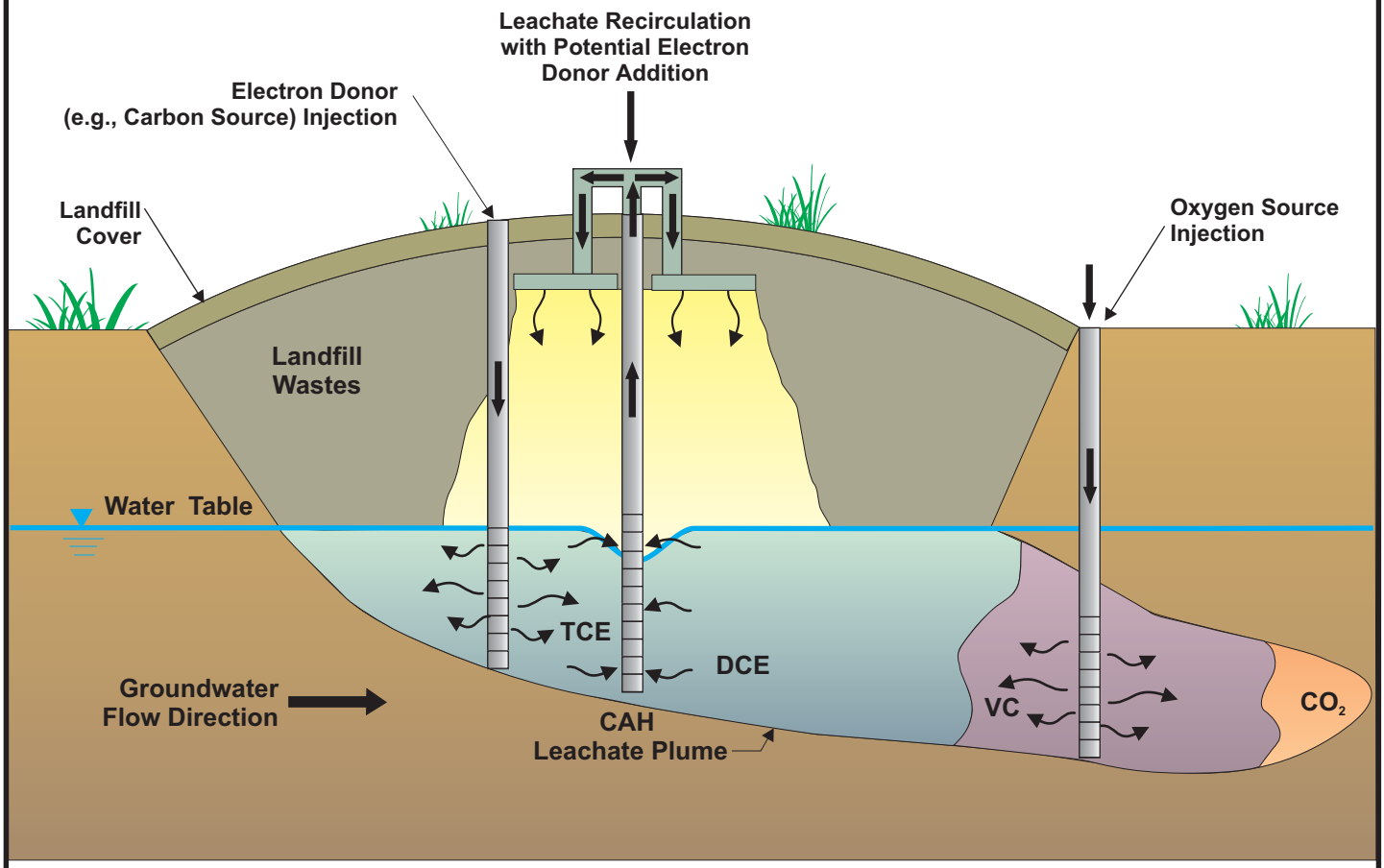

FIGURE 4.10

CREATION OF SEQUENTIAL ENHANCED ANAEROBIC AND AEROBIC ZONES BENEATH A LANDFILL

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Available data suggest that the groundwater pH at many USAF landfills is within the range of 6.0 to 8.0; therefore, pH adjustment may not be necessary or desired unless it becomes a limiting factor inhibiting the maintenance of the methanogenic conditions conducive to optimal biodegradation of CAHs.

In order to avoid inhibition of methane-fermenting organisms, the pH should be maintained above 6.0 standard units (Emcon, 1982), and even higher levels (7.0 to 7.8) should be sustained to provide optimal conditions. According to Harper and Just (1995), studies have shown that addition of pH-altering amendments is not required once methanogenesis is active because acids are consumed in a self-balancing system. However, addition of caustics to recirculated water may be required to stimulate methanogenesis. Johnston *et al.* (1996) report that the natural pH buffering provided by calcium-carbonate-bearing formations may enhance methanogenic biodegradation.

Tittlebaum (1982) describes the construction of four simulated landfills to study the feasibility of a leachate recirculation system in providing leachate treatment and control of groundwater pollution, and to examine the effect of leachate pH and nutrient control on both shredded and unshredded refuse with regard to accelerated rates of biological stabilization. The study concluded that 1) recirculation with enhancement by neutralization for pH control (at 7.0 standard units) more rapidly establishes an active anaerobic biological population within the fill; and 2) pH control of leachate leads to an increased rate of biological stabilization with significant reductions of BOD, COD, and total organic carbon (TOC).

Pohland *et al.* (1993) describe a near-full-scale demonstration project during which one of six landfill cells was operated using leachate recycling. Study results illustrated the benefits of pH and moisture control and the increased gas volume produced by the implementation of leachate containment, collection, and recirculation.

Addition of Nutrients. In addition to proper electron donor selection, nutrient availability may be a critical factor in maintaining a healthy dechlorinating microbial population. However, based on the literature review performed for this project and summarized in Section 3, nutrient addition to promote biodegradation of chlorinated solvents appears to be relatively rare. A study conducted by Tittlebaum (1982) that utilized simulated landfills to examine the effect of leachate nutrient control on both shredded and unshredded refuse with regard to accelerated rate of biological stabilization concluded that nutrient control is not necessary or of any advantage. In contrast, Lee *et al.* (1995) state that the low concentrations of phosphorus (< 1 mg/L) may, in addition to other factors, control microbial activity at the DuPont Necco Park Landfill in New York, which is contaminated with numerous CAHs at concentrations up to hundreds of mg/L. Similarly, Harper and Just (1995) list phosphorous addition to recirculated leachate (nitrogen is typically abundant) as one of the strategies that has proven successful for stimulating methanogenic activity in landfill environments.

Based on these observations, evaluation of phosphorus concentrations in leachate-impacted groundwater is recommended. In this way, a determination can be made as to whether concentrations of this parameter are likely to limit microbial activity within and beneath the landfill.

4.2.4.6 Containment and Collection of Landfill Gas

The rate at which municipal waste generates gas increases for the first 5 or 6 years after waste placement, and then declines as the amount of biodegradable organic matter decreases (Weand *et al*., 1999). Most DoD landfills are more than 20 years old; therefore, they are likely to produce only small amounts of landfill gas after cover placement, and design/installation of specialized gas-collection systems will probably not be necessary in most instances. Alternative covers that do not include barrier layers may not need gas control because any produced gases would migrate harmlessly into the atmosphere. This topic is discussed further in Section 4.3.

4.2.4.7 Cost of Leachate Recirculation and Bioreactors

Costs for leachate recirculation and bioreactor systems applied to unlined landfills are not available. Limited cost information is available for leachate recirculation and bioreactors installed in fully-lined commercial domestic landfill operations. We have used these costs, and added typical costs for extraction well/trench collection systems to develop a total system cost estimate for collection and recirculation of leachate-contaminated groundwater and bioreactor construction and operation at unlined landfills.

At many sites contaminated with CAHs, leachate collection is a "fixed" cost that would apply to any remedial design. It is our experience that even if a conventional RCRA cap is installed over an unlined landfill, there will generally be a requirement for some level of leachate collection if a CAH plume has formed. The exceptions to this would be at arid sites with deep, unimpacted groundwater, and sites where a MNA alternative has been accepted. A capital cost of $150 per linear foot (LF) of landfill perimeter was estimated for collection trench and sump-pump installation (Means, 2001). This cost assumes a 25-foot trench depth, an 8-foot saturated thickness, and a landfill area of 30 square acres with leachate collection on 3 sides (3,400 LF of trench or 113 LF/acre at $150/LF = $17,000/acre). Higher costs would be incurred with a deeper trench and thicker saturated interval. As discussed in Section 4.2.2, leachate recirculation systems are best suited for lower permeability, shallow aquifers with relatively thin saturated intervals.

Collected leachate would be transferred directly to the leachate recirculation manifold on top of the bioreactor. Studies of leachate recirculation systems at lined commercial landfills have indicated average capital costs for recirculation of approximately $40,000 to $50,000 per acre (Hater, 2001).

The cost of a four-foot-thick, compost/mulch sublayer constructed above a CAH source area without removing existing waste would be $40,000 to $70,000 per acre. Therefore, the total cost per acre for leachate collection, leachate recirculation, and bioreactor construction would be approximately $100,000 to $140,000 per acre.

Operating costs for the recirculation/bioreactor system would include electricity to operate pumps and other O&M activities such as weekly system maintenance and piezometer readings, and monthly analysis of leachate for pH, ORP, and CAHs. These O&M costs have been estimated at $60,000 to $80,000 per year by the authors, and $92,000 per year for a lined landfill recirculation system (Hater, 2001).

A full-scale bioreactor cell comprising 8 acres was constructed at a landfill in Augusta, Georgia. Leachate recirculation is combined with air injection. Results of a 6-month pilot project indicate that the cost for the bioreactor is approximately $35,000-$50,000/acre with an O&M cost of $25,000/month (Magnuson, 1998).

Pohland *et al*. (2001) list bioreactor developments at the Delaware Solid Waste Authority as well as descriptions of selected pilot- and full-scale bioreactor landfills reported in the technical literature. Pertinent cost elements provided in this summary include:

- Bioreactor construction and operating costs of $1,035,000 and $450,000, respectively, for a 7-acre landfill in Idaho. Leachate recirculation was accomplished using a combination of surface spray, trenches, and wells.

- Installation of 11 leachate-distribution trenches filled with shredded tires at an approximately 13-acre landfill in Minnesota at a cost of $290,000. Implementation of this approach reportedly resulted in a savings of approximately $72,500 per year during the 1997-1998 time frame compared to hauling and offsite disposal of leachate.

According to Reinhart *et al*. (2001), full-scale operating bioreactors report annual cost savings varying from $75,000 to $500,000. However, Gambelin and Cochrane (1998) developed a life cycle analysis that estimated a cost differential of $1.40 to $2.15/ton in favor of dry landfills. The discrepancy between these estimates highlights the difficulties in generalizing bioreactor costs due to the large number of site-specific variables.

4.3 LANDFILL COVER DESIGNS

There are three primary reasons for constructing landfill covers (Weand *et al*., 1999):

- Controlling infiltration and resultant leachate formation;

- Isolating wastes from direct contact with potential receptors at the surface; and

- Controlling release of explosive or toxic landfill gases.

As described in Section 1.3.2.2, both MSW and HW final-cover regulatory specifications currently include specific permeability requirements. Final covers for HW landfills are to be constructed to minimize the migration of liquids through the "closed" landfill, by providing a permeability less than or equal to the bottom liner or natural subsoils. Similarly, for MSW landfills, the general goal of the "closure" cover is to minimize infiltration.

Weand *et al*. (1999) describe characteristics of USAF landfills in particular, and DoD landfills in general, that set them apart from most currently operating non-military landfills and influence selection of an appropriate cover type. These characteristics include:

- Military landfills generally do not have a bottom liner;

- Most USAF landfills have not been operational since the late 1980s, and much of the waste has already consolidated and decomposed, minimizing the potential for future subsidence; and

- Landfill gas production can be expected to be low, potentially obviating the need for a gas-collection/control system.

4.3.1 Landfill Cover Types and Applicability

Three general types of covers are recommended for consideration at DoD landfills, depending on site-specific characteristics. These cover types, including low-permeability, ET, and capillary-barrier covers, are briefly described below. Other types of covers exist; however, the three aforementioned cover types are those that are deemed to potentially have broad applicability at military landfills. Excellent, in-depth discussions of landfill cover selection and design that focus on alternatives to impermeable caps are contained in Weand *et al*. (1999), Hauser *et al*. (2001a), and Suthersan (2001).

4.3.1.1 Low-Permeability Cover

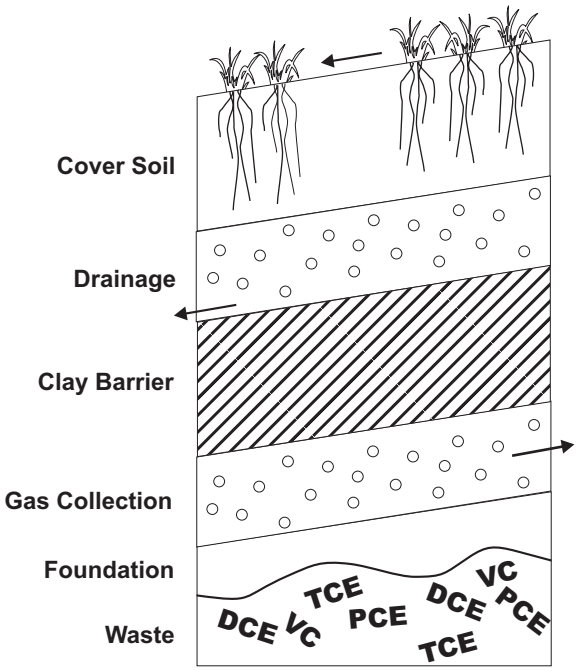
There are many different configurations of conventional, barrier-type landfill covers. The components of multi-layer impermeable landfill caps usually include a vegetative soil layer at the land surface, overlying a drainage layer, overlying a barrier layer above the waste materials (Harper and Just, 1995). Weand *et al*. (1999) and Hauser *et al*. (2001a) report that a RCRA Subtitle C (HW landfill) cover typically contains a permeable gas-collection layer beneath the barrier layer, which is underlain by a foundation layer that separates the cover from the waste and provides the correct surface grade. A schematic diagram of the various layers of a low-permeability RCRA Subtitle C cover is shown on Figure 4.11. Construction costs for conventional covers constructed at eight USAF landfills in Illinois, Mississippi, Texas, and New Hampshire ranged from $319,000 to $571,000 per acre of landfill.

4.3.1.2 ET Cover

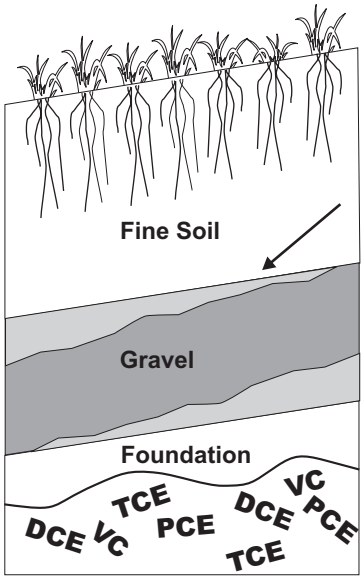
In its simplest form, an ET cover is a vegetated soil cover with a sufficiently deep soil profile to allow storage of infiltrated water until its removal by evaporative losses from the soil surface and by uptake into plant roots at depth in the profile (Suthersan, 2001). Weand *et al*. (1999) state that the ET cover is an inexpensive, practical, and easily maintained biological system what will remain effective over extended periods of time. Suthersan (2001) uses the term “phyto-cover” and describes it as a “protective, living ‘skin’ for a landfill…that can equal the percolation-blocking performance of a RCRA cap while being substantially more cost-effective and providing additional benefits.” An ET cover is shown schematically on Figure 4.11.

Costs for ET covers have been compared to conventional RCRA Subpart D covers based on actual costs from military installations and from cost models developed for sites in the southern Great Plains of North America. Based on a survey of conventional RCRA covers installed on 8 military landfills, the cost per acre ranged from $319,000 to $571,000 per acre of landfill (Hauser *et al*., 1999). A recently constructed ET cover at F.E. Warren AFB, WY cost approximately $150,000 per acre to construct. A cost model

Conventional Cover



Capillary Barrier



ET Cover

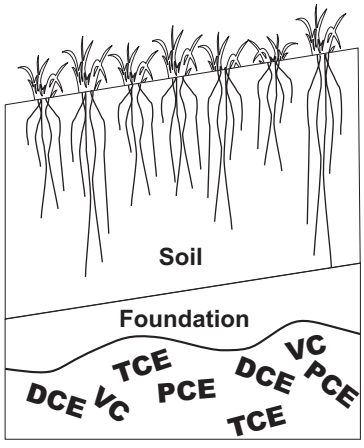


FIGURE 4.11

SELECTED LANDFILL COVER TYPES

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developed for the southern Great Plains of North America estimated the cost of ET covers, without gas collection, at $176,000 per acre (Hauser *et al*., 2001b). Based on these actual and estimated costs, the capital cost for ET covers is generally one-half to one-third the cost of conventional RCRA Subpart D covers. ET covers will be most cost effective in semi-arid environments where precipitation does not exceed evapotranspiration rates and significant clay sub-layers are not required beneath the vegetated top cover. Hauser *et al*. (2001a) also state that a cost savings of $150,000 per acre of cover is a conservative figure appropriate for general use. Hauser *et al*. (2001a) estimate that it would cost 60 to 65 percent less to repair 100- and 400-foot-long cracks in an ET cover than in a conventional barrier-type cover.

4.3.1.3 Capillary-Barrier Cover

This type of cover includes a relatively fine soil layer overlying a relatively coarse layer (Figure 4.11). The resulting layering is a "capillary barrier" because the coarse soil layer does not have sufficient matric potential to "pull" moisture from the high-capillarity soil above. The fine-grained layer does not necessarily have a low conductivity. Rather, it is the coarse soil below that impedes soil-water flow. Suthersan (2001) reports that this type of cover can be thought of as an enhanced ET cover because it relies on water removal via ET, but is designed such that water storage near the surface (in the fine-grained soil layer) is enhanced to promote the efficient removal of infiltrated water via ET. Construction cost estimates for capillary barriers were not encountered in the literature.

4.3.2 Landfill Cover Selection Decision Tree

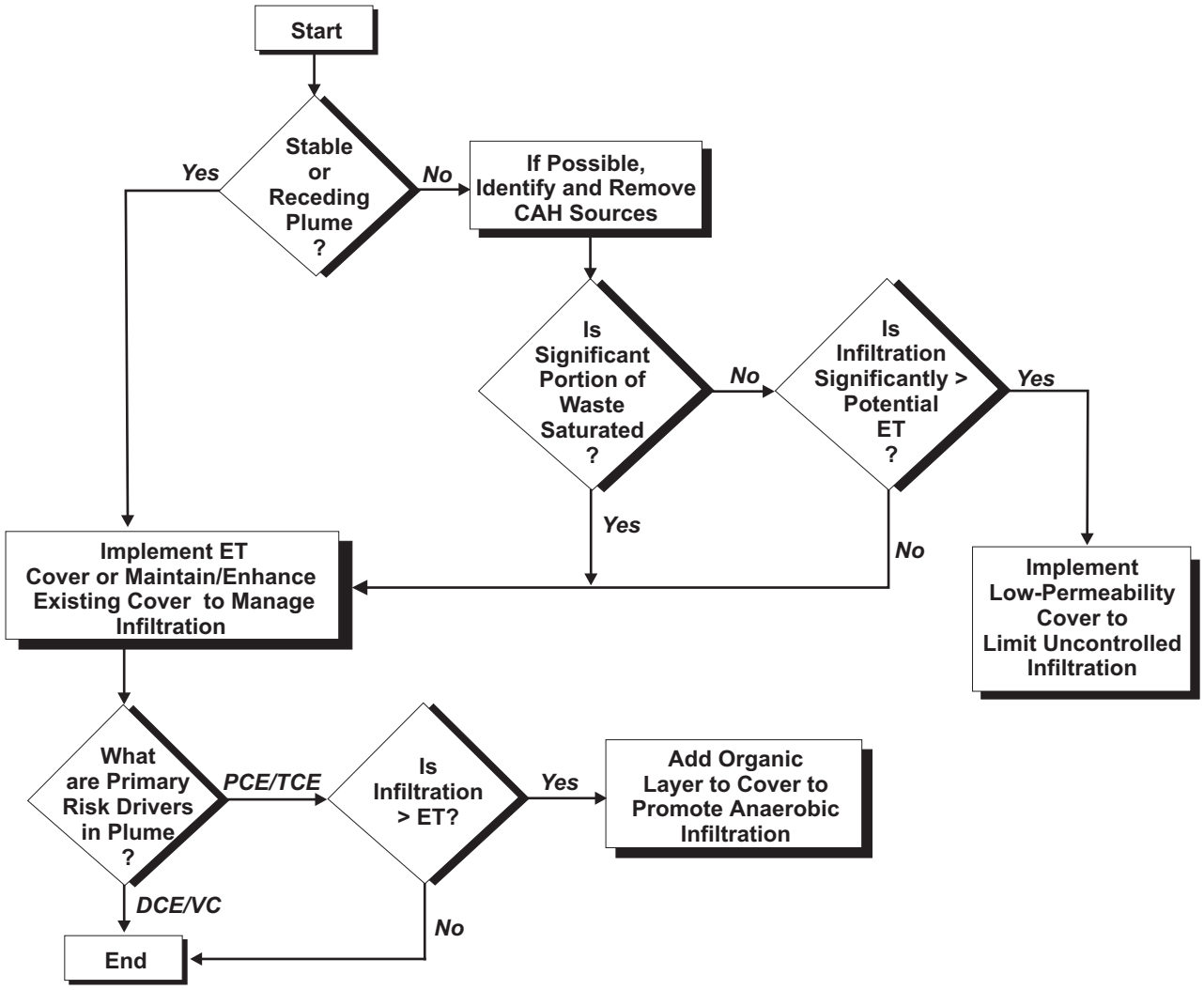
The site characteristics that have a dominant influence on choosing an appropriate final cover include climate, soils, landfill characteristics, hydrogeology, gas production, seismic environment, and reuse of landfill areas. A decision tree for selecting an appropriate cover for CAH-contaminated landfills is depicted on Figure 4.12, and is described in the following paragraphs.

The dynamics of the CAH plume should be assessed prior to cap construction (Figure 4.1). The presence of an expanding plume indicates that significant CAH source mass is present in the landfill. Identification and removal of as much source material as possible prior to final cover installation would reduce the persistence and migration potential of the groundwater plume. As described in Section 4.2.1, source identification can be accomplished using a combination of historical research, geophysical field techniques to locate buried metallic objects, and soil gas sampling to identify contamination "hotspots", followed by confirmation test pits or soil borings.

4.3.2.1 Conditions Conducive to Low-Permeability Covers

It is appropriate to implement a low-permeability cover to limit infiltration if either of the following conditions exist:

- The contaminant source material generally remains above the seasonally high water table, or



Notes:
CAH = Chlorinated aliphatic hydrocarbon
DCE = Dichloroethene
ET = Evapotranspiration
PCE = Tetrachloroethene
TCE = Trichloroethene
VC = Vinyl Chloride

FIGURE 4.12

LANDFILL COVER DESIGN DECISION TREE

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- The climate is characterized by abundant rainfall, and the capacity of ET covers and capillary barriers to store water and release it back to the atmosphere may be overwhelmed by precipitation amounts. In this situation, it would be difficult to control infiltration into the landfill, and the field capacity of the landfill may be exceeded, potentially causing higher-than-desired leachate production rates, accelerated plume migration, and/or surface leachate seeps. This situation also could result in uncontrolled expansion of a CAH plume underlying the landfill.

Low-permeability caps function as vapor and moisture barriers that can minimize the infiltration of atmospheric oxygen and oxygenated precipitation/runoff into the landfill. Therefore, this type of cover can facilitate enhancement or maintenance of anaerobic conditions that are conducive to CAH biodegradation. However, they generally will not allow sufficient infiltration of moisture to sustain microbial processes at the desired rate. Therefore, installation of a low-permeability cover combined with subsurface injection of recirculated leachate-contaminated groundwater can satisfy both objectives (i.e., minimal infiltration of atmospheric oxygen/oxygenated precipitation, and maintenance of sufficient moisture levels). As discussed in Section 4.3.2.4, subsurface anaerobic conditions can also be maintained or enhanced using alternative cover types (e.g., ET covers) combined with an organic layer to promote anaerobic infiltration. This latter option may be substantially more cost-effective than installation of a low-permeability cover, depending on site-specific factors such as the type of cover required and the local availability of fine-grained soils that could be used to construct the landfill cover.

4.3.2.2 Conditions Conducive to ET-Type Covers

If a significant portion of the contaminant source material is at least seasonally saturated (i.e., resides below the seasonally high water table), or if precipitation rates are such that the infiltration rate can be sufficiently controlled without the addition of a low-permeability cap, then an ET-type cover can be installed to manage infiltration. If at least a portion of the CAH source lies within the saturated zone, then installation of a low-permeability cap likely will have little influence on reduction of contaminant leaching rates. The presence of a thick vadose zone with substantial water storage capacity is conducive to use of ET covers that may allow some infiltration to occur. Climate is a primary determinant as to whether or not an ET cover is practical for a given site. Weand *et al*. (1999) estimate that properly designed ET covers could prevent infiltration into landfill wastes in most of the US west of the Mississippi River, and could minimize infiltration at numerous locations in much of the rest of the country.

If the soil between the landfill and the groundwater table is of low permeability, or a contiguous low-permeability layer exists, then ET covers could be considered even in wetter climates. In general, soils with hydraulic conductivities of less than 10^{-7} centimeters per second (cm/sec) will probably serve as effective barriers to infiltration.

Two conditions may promote anaerobic degradation of DCE and VC short of methanogenesis: 1) VC can directly oxidize to carbon dioxide (CO_2) under iron(III)-reducing conditions; and 2) DCE can degrade anaerobically to CO_2 under manganese(IV)-reducing conditions (Bradley, 2000). Therefore, one may infer that soils with sufficient organic content, and sufficient iron and manganese oxides may promote

more efficient anaerobic degradation of DCE and VC. For such conditions, ET covers might be given extra consideration when DCE or VC is a COC.

Finally, soils with very high organic matter are candidates for ET covers, even in moderate climates, because of the retardation of organic-contaminant migration velocities. Organic matter fractions of greater than 0.01 will cause significant retardation and greatly increase the time available for contaminant degradation prior to reaching a potential receptor exposure point.

In summary, ET covers should be considered when it is desirable to eliminate or reduce infiltration of water. ET covers may control infiltration of water as effectively as more expensive low-permeability covers in semi-arid or arid climates, where ET rates are similar to or greater than the value of precipitation (minus runoff) on average. In wetter regions, ET covers can potentially reduce infiltration sufficiently to achieve remedial goals. However, based on the work of Warren *et al*. (1996b), Gill *et al*. (1999), and Weand *et al*. (1999), ET covers can fail during rainy periods, even in arid regions. This may be particularly true for many arid regions in the US, which have summer monsoon seasons. These observations indicate the need to evaluate the most critical seasonal precipitation event during design (e.g., snowmelt and early spring rain). Hauser *et al*. (2001a) concluded that the following factors may be at least partially responsible for past failures of ET covers:

- Inadequate soil depth; and/or
- Over-compaction of soils resulting in reduced water-holding capacity and/or poor root growth.

Therefore, proper and informed selection, design, and inspection of ET covers greatly increases chances for success.

In most cases, conventional ET covers are recommended over capillary barriers. This conclusion is based on the fact that capillary barriers have been used primarily in experimental installations, and field experience shows that they may fail periodically (Nyhan *et al*., 1990; Warren *et al*., 1996a). Based on the research of Warren *et al*. (1996b), even in semi-arid climates, the ability of capillary barriers to shed water laterally along the capillary break within covers is likely to be inadequate, particularly during months with frequent rainfall. This would be particularly true in areas with relatively thin vadose zones (e.g., near rivers or washes). Infiltration reduction may be improved by incorporating a capillary barrier with the ET cover design.

4.3.2.3 Use of the Existing Soil Cover

In some cases, maintenance of the existing soil cover may be adequate to achieve remedial goals. Examples of such sites include relatively uncontaminated landfills, contaminated landfills with very thick vadose zones (e.g., in arid areas) where infiltration of precipitation will not result in significant groundwater contamination, and old landfills where past leaching and biodegradation have stabilized the waste mass. In all cases, the existing soil cover must adequately isolate the waste from potential receptors and promote proper surface drainage (e.g., prevent ponding). An existing cover that is

maintained or enhanced to prevent erosion and promote runoff and healthy vegetation can function as a surrogate ET cover.

4.3.2.4 Inclusion of Organic Sub-Layers in Landfill Covers

Installation of a landfill cover typically reduces the migration of oxygen into the landfill by reducing water and air infiltration. Thus, landfill covers promote anaerobic conditions in the interior of the landfill. As described in Section 4.2.4.5, promotion of anaerobic conditions is desirable in many, if not most landfills with CAH contamination because common highly chlorinated solvents such as PCE and TCE can be most efficiently biodegraded in such environments.

The recent installation of a horizontal bark-mulch layer at the ground surface and a vertical bark-mulch wall in the saturated zone at Offutt AFB, Nebraska (Haas *et al.*, 2000) indicates that percolation of surface water or groundwater through an organic layer can scavenge available oxygen and enhance reductive dechlorination of CAHs in groundwater. The limited experience with this approach to date suggests the utility of adding an organic (e.g., mulch) layer to a landfill cover if the primary risk drivers are highly-chlorinated solvents such as PCE and TCE. The most likely use of this type of layer is in ET-type covers that control, but do not completely eliminate, water infiltration through the cover. Such a sub-layer could be incorporated into the landfill cover beneath the surficial soil layer, and could potentially comprise the foundation that separates the waste mass from the soil cover (Figure 4.13)**.** The length of time that an organic layer will enhance the organic carbon content and scavenge available DO from percolating recharge water is not known at this time, nor is the optimal thickness of the organic layer.

Organic material also could be used to amend the chemistry of recirculated leachate-contaminated groundwater by passing the leachate through reactive beds of organic material prior to reinjecting it into the landfill. Alternatively, recirculation trenches could be lined with organic material to optimize the geochemistry of the influent water and enhance anaerobic conditions within the landfill.

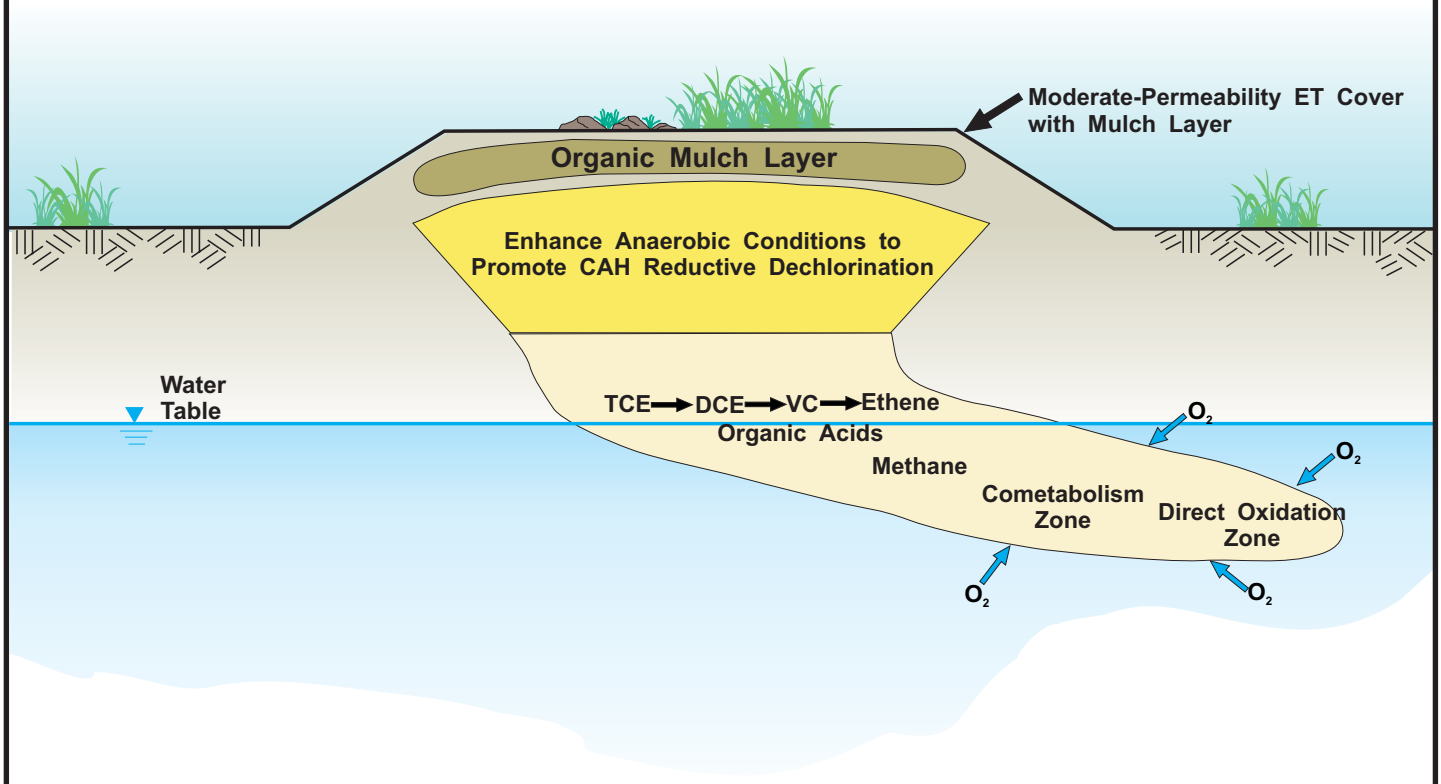

FIGURE 4.13

USE OF ORGANIC LAYER TO ENHANCE REDUCTIVE DECHLORINATION OF CAHs

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SECTION 5

RECOMMENDATIONS

Section 5 has been removed from this abbreviated version of the report. Please contact John Hicks (john.hicks@parsons.com) or Doug Downey (doug.downey@parsons.com) at Parsons in Denver, Colorado (303-831-8100) to obtain a copy of this section.

SECTION 6

REFERENCES

Albaiges, J. (other authors unknown). 1986. Organic Indicators of groundwater pollution by a sanitary landfill. *Water Res*. 20(9): 1153.

Albrechtsen, H., P.L.Bjerg, L. Ludvigsen, K. Rugge, and T.H. Christensen. 1999. An anaerobic field injection experiment in a landfill leachate plume, Grindsted, Denmark. *Water Resources Research* 35(4): 1247-1256. April.

Anderson, R.T., and D.R. Lovley. 1997. Ecology and biogeochemistry of in situ groundwater remediation. *Advances in Microbial Ecology*, Volume 15. Plenum Press: New York, NY.

Bagchi, A. 1990. *Design, Construction, and Monitoring of Sanitary Landfills*. John Wiley & Sons, Inc.: Canada.

Bagley, D.M., and J.M. Gossett. 1989. Tetrachloroethene transformation to trichloroethene and cis-1,2-dichloroethene by sulfate-reducing enrichment culture. *Applied and Environmental Microbiology* 56(8):2511-2516.

Barker, J.F., J.S. Tessmann, P.E. Plotz, and M. Reinhard. 1986. The Organic Geochemistry of a Sanitary Landfill Leachate Plume. *Journal of Contaminant Hydrology* 1: 171-189.

Barlaz, M.A., and M.A. Gabr. 2000. Closing the gap in the regulation of municipal solid waste landfills: Defining the end of the post closure monitoring period and the future stability of leachate recirculation landfills. North Carolina State University Web Site: www.4.ncsu.edu:8030/~barlaz/post_closure/menu_txt.html.

Barr, G.L. and M. Fernandez Jr.. 1980. Water-Quality Data from a Landfill-Leachate Treatment and Disposal Site, Pinellas County, Florida. January 1979-August 1980. US Geological Survey Open-File Report 81-1064.

Barr, K.D., B.D. O'Flanagan, W.A. Newman, J.K. Julik, and D.W. Wetzstein. 1997. In Situ Treatment of Landfill Leachate Using Bioventing. *Waste Age* 28(1): 81-87. January.

Basberg, L., D. Banks, and O.M. Saether. 1998. Redox Processes in Groundwater Impacted by Landfill Leachate. *Aquatic Chemistry* 4: 253-272.

Becvar, E.C., C. Vogel, G. Sewell, J. Gossett, and S. Zinder. 1997. In situ dechlorination of solvents in saturated soils. *In Situ and On-Site Bioremediation.* Volume 3. Battelle Press: Columbus, Ohio.

Beeman, R.E. 1994. A field evaluation of in situ microbial reductive dehalogenation by the biotransformation of chlorinated ethenes. *Bioremediation of Chlorinated and Polycyclic Aromatic Hydrocarbon Compounds*. Lewis Publishers: Boca Raton, FL.

Bendz, D., V.P. Singh, and M. Åkesson. 1997. Accumulation of water and generation of leachate in a young landfill. *Journal of Hydrology* 203:1-10.

Berwanger, D.J. and J.F. Barker. 1988. Aerobic Biodegradation of Aromatic and Chlorinated Hydrocarbons Commonly Detected in Landfill Leachates. *Water Poll. Res. J. Canada* 23(3): 460-474.

Bjerg, P.L., K. Riigge, J. Cortsen, P.H. Nielsen, and T.H. Christensen. 1999. Degradation of aromatic and chlorinated aliphatic hydrocarbons in the anaerobic part of the Grindsted landfill leachate plume: in situ microcosm and laboratory batch experiments. *Ground Water* 37(1):113-121.

Bouwer, E.J. 1994. Bioremediation of chlorinated solvents using alternate electron acceptors. *Handbook of Bioremediation*. Lewis Publishers, Inc.: Boca Raton, FL.

Boyer, I., V. Hauser, D.M. Gimon, and M.D. Gill. 1999. *Decision Tool for Landfill Remediation*. Prepared for Air Force Center for Environmental Excellence. Mitretek Systems. August.

Bradley, P.M. 2000. Redox processes in groundwater impacted by landfill leachate. *Aquatic Geochemistry* 4:253-272, 1998.

Bradley, P.M., and F.H. Chapelle. 1996. Anaerobic mineralization of vinyl chloride in Fe(III)-reducing aquifer systems. *Environmental Science and Technology* 30(6):2084-2086.

Bradley, P.M., and F.H. Chapelle. 1997. Kinetics of DCE and VC mineralization under methanogenic and Fe(III)-reducing conditions. *Environmental Science and Technology* 40:2084-2086.

Bradley, P.M., and F.H. Chapelle. 1998. Effect of contaminant concentration on aerobic microbial mineralization of DCE and VC in stream-bed sediments. *Environmental Science and Technology* 32(5):553-557.

Brasaemle, J.E., Y. Lin, and D.W. Marczely. 1997. *Natural Attenuation of Chlorinated Compounds at a Closed Municipal Landfill*. *In Situ and On-Site Bioremediation*, Volume 3. Battelle Press: Columbus, OH.

Brigmon, R.L., C.B. Fliermans, M.M. Franck, D. Altman, F. Washburn, C. Berry, A. Glucksman, and K. Lombard. 1997 Intrinsic remediation of landfills. *In Situ and On-Site Bioremediation*, Volume 3. Battelle Press: Columbus, OH.

Brigmon, R.L., D.J. Altman, M.M. Franck, C.B. Fliermans, T.C. Hazen, and A.W. Bourquin. 1999. Evaluation of Methanogenic Bacteria During Injection of Gaseous Nutrients for *In Situ* Trichloroethylene Bioremediation in a Sanitary Landfill. In *Engineered Approaches for In Situ Bioremediation of Chlorinated Solvent Contamination, Proceedings of the Fifth International In Situ and On-Site Bioremediation Symposium*. April. Battelle Press.

Buchanan, Jr., R.J., E.J. Lutz, D.E. Ellis, C.L. Bartlett, G.J. Hanson, M.D. Lee, M.A. Heitkamp, M.A. Harkness, K.A. DeWeerd, J.L. Spivak, J.W. Davis, G.M. Klecka, and D.L. Pardieck. 1997. Anaerobic reductive dehalogenation pilot design for Dover AFB. *In Situ and On-Site Bioremediation*, Volume 3. Battelle Press: Columbus, OH.

Carter, S.R., and W.J. Jewell. 1993. Biotransformation of tetrachloroethylene by anaerobic attached-films at low temperatures. *Water Research* 27. April.

Chen, C., J.A. Puhakka, and J.F. Ferguson. 1996. Transformation of 1,1,2,2-tetrachloroethane under methanogenic conditions. *Environmental Science and Technology* 30:542-547.

Cherry, J.A. 1990. Groundwater monitoring: some deficiencies and opportunities. In *Hazardous Waste Site Investigations: Towards Better Decisions, Proceedings of the 10th ORNL Life Sciences Symposium.* Lewis Publishers: Gatlinburg, TN.

Chian, E.S.K. 1977. Stability of organic matter in landfill leachates. *Water Res*. 11:225.

Chian, E.S.K., and F.B. DeWalle. 1976. Sanitary landfill leachates and their treatment. *Journal of the Environmental Engineering Division* 102:411-431.

Christensen, T.H, P. Kjeldsen, H.J. Albrechtsen, G. Heron, P.H. Nielsen, P.L. Bjerg, and P.E. Holm. 1994. Attenuation of landfill leachate pollutants in aquifers. *Critical Reviews in Environmental Science and Technology* 24(2):119-202.

Christensen, T.H., P.L Bjerg, and P. Kjeldsen. 2000. Natural Attenuation: A Feasible Approach to Remediation of Ground Water Pollution at Landfills? *Ground Water Monitoring & Remediation*, Winter, pp. 69-77.

Chu, L.M., K.C. Cheung, M.H, Wong. 1994. Variations in the chemical properties of landfill leachate. *Environmental Management* 18(1):105-117.

Davis, J.W., and C.L. Carpenter. 1990. Aerobic biodegradation of vinyl chloride in groundwater samples. *Applied Environmental Microbiology* 56:3878.

de Bont, J.A.M., M.J.W. Vorage, S. Hartmans, and W.J.J. van den Twell. 1986. Microbial degradation of 1,3-dichlorobenzene. *Applied and Environmental Microbiology* 52:677-680.

DeBruin, W.P., M.J.J. Kotterman, M.A. Posthumus, G. Schraa, and A.J.B. Zehnder. 1992. Complete biological reductive transformation of tetrachloroethene to ethane. *Applied and Environmental Microbiology* 58:1996- 2000.

Deipser, A. and R. Stegmann. 1994. The origin and fate of volatile trace components in municipal solid waste landfills. *Waste Management & Research* 12: 129-139.

Downey, D.C., R.E. Hinchee, and R.N. Miller. 1999. Cost-Effective Remediation and Closure of Petroleum-Contaminated Sites. Battelle Press. 297 pp.

Edwards, E.A., and E.E. Cox. 1997. Field and laboratory evidence of sequential aerobic chlorinated solvent biodegradation. *In situ and On Site Bioreclamation*, Volume 3. Battelle Press: Columbus, OH.

Eganhouse, R.P., I.M. Cozzarelli, M.A. Scholl, and L.L. Matthews. 2001. Natural attenuation of volatile organic compounds (VOCs) in the leachate plume of a municipal landfill: using alkylbenzenes as process probes. *Ground Water* 39(2):192-202). March/April.

El-Fadel, M. 1999. Leachate recirculation effects on settlement and biodegradation rates in msw landfills. *Environmental Technology* 20:121-133.

Ellis, B., K Gorder, D. Rees, and J.F. Rees. 1999. Intrinsic Remediation of Chlorinated Solvents at a Closed Industrial Landfill in the UK. In *Natural Attenuation of Chlorinated Solvents, Petroleum Hydrocarbons, and Other Organic Compounds, Proceedings of the Fifth International In Situ and On-Site Bioremediation Symposium*, San Diego, CA. April 1999. Battelle Press.

EMCON Associates. 1982. Methane Generation and Recovery from Landfills. Ann Arbor Science Publishers, Inc. Ann Arbor, Michigan.

Environmental Simulations, Inc. 2000. Guide to Using WINFLOW, Version 2.

Fathepure, B.Z., J.P. Nengu, and S.A Boyd. 1987. Anaerobic bacteria that dechlorinate perchloroethene. *Applied and Environmental Microbiology* 53:2771-2674.

Federal Remediation and Treatment Roundtable (FRTR). 2000. Remediation Technologies Screening Matrix and Reference Guide. http://www.frtr.gov/ matrix2.

Fennell, D.E., and J.M. Gossett. 1997. Comparison of bytyric acid, ethanol, lactic acid, and propionic acid as hydrogen doors for the reductive dechlorination of trichlorothene. *Environmental Science and Technology* 31:918-925.

Fetter, C.W. 1999. *Contaminant Hydrogeology*. Second edition. Prentice Hall: Upper Saddle River, NJ.

Freedman, D.L., and J.M. Gossett. 1989. Biological reductive dechlorination of tetrachloroethylene and trichloroethylene to ethylene under methanogenic conditions. *Applied and Environmental Microbiology* 55:2144-2151.

Gambelin, D. J. & D. Chochrane (1998) Life Cycle analysis of a Bioreactor landfill in California. Proceedings from SWANA's 3rd Annual Landfill Symposium, Palm Beach Gardens, FL, June 22-24.

Gabr, M.A., M.S. Hossin, M.A. and Barlaz. 2000. Solid waste settlement in landfills with leachate recirculation. *Geotechnical News* 18(2).

Gerritse, J. V. Renard, J. Visser, J.C. Gottschal. 1995. Complete degradation of trichloroethene by combining anaerobic dechlorinating and aerobic methanetrophic enrichment cultures. *Applied Microbiology and Biotechnology* 43:920.

Gerritse, J., O. Drzyzga, G. Kloetstra, M. Keijmel, L.P. Wiersum, R. Hutson, M.D. Collins, and J.C. Gottschal. 1999. Influence of different electron donors and acceptors on dehalospiration of tetrachloroethene by *Desulfitobacterium frappiere* TCE1. *Applied and Environmental Microbiology* 65:5212-5221.

Gerritse, J., V. Renard, T.M. Pedro Gomes, P.A. Lawson, M.D. Collins, and J.C. Gotschal. 1996. *Desulfitobacterium* sp. strain PCE1, an anaerobic bacterium that can grow by reductive dechlorination of tetrachloroethene or ortho-chlorinated phenols. *Archives of Microbiology* 165:132-140.

Gibson, S.A., and G.W. Sewell. 1992. Stimulation of reduction dechlorination of tetrachloroethene in anaerobic aquifer microcosms by addition of short chain organic acids or alcohols. *Applied and Environmental Microbiology* 58:1392-1393.

Gill, M.D., V.L. Hauser, J.D. Horin, B.L. Weand, and D.J. Casagrande. 1999. *Landfill Remediation Project Manager's Handbook.* Air Force Center for Environmental Excellence. Prepared by Mitretek Systems for Air Force Center for Environmental Excellence. Brooks AFB. December.

Gillham, R.W., and S.F. O'Hannesin. 1994. Enhanced degradation of halogenated aliphatics by zero-valent iron. *Ground Water* 32(6):958-967.

Goltz, M.N., G.D. Hopkins, J.P. Allan, M.E. Dolan, and P.L. McCarty. 1997. Full-scale demonstration of in situ aerobic cometabolism of trichloroethylene-contaminated groundwater. *In Situ and On-Site Bioremediation,* Volume 3. Battelle Press: Columbus, OH.

Gossett, J.M., and S.H. Zinder. 1996. Microbiological aspects relevant to natural attenuation of chlorinated ethenes. In *Proceedings of the Symposium on Natural*

Attenuation of Chlorinated Organics in Groundwater. Dallas, TX. EPA 540/R-96/509. September 11-13.

Gossett, J.M., and S.H. Zinder. 1997. Microbiological aspects relevant to natural attenuation of chlorinated ethenes. In *Proceedings, US EPA Symposium in Natural Attenuation of Chlorinated Organics in Ground Water*. Dallas, TX. EPA/540/R-97/504/1997. September.

Guadagnini, A., and S. Franzetti. 1999. Time related capture zones for contaminants in randomly heterogeneous formation. *Ground Water* 37(2):253.

Haas, P., P. Cork, and C.E. Aziz 2000. In Situ Biowall Containing Organic Mulch Promotes Chlorinated Solvent Bioremediation. In *Proceedings of the Second International Conference on Remediation of Chlorinated and Recalcitrant Compounds*, Monterey, CA. May 2000. Volume C2-4, p 71-76.

Hanson, R.S., and T.E. Hanson. 1996. Methanotrophic bacteria. *Microbiology Review* 60(2):439-471.

Harkness, M.R., A.A. Bracco,.M.J. Brennan, K.A. Deweerd, and J.L. Spivack. 1999. Use of bioaugmentation to stimulate complete reductive dechlorination of trichloroethene in Dover soil columns. *Environmental Science and Technology* 33:1100-1109.

Harmsen, J. 1983. Identification of organic compounds in leachate from a waste tip. *Waste Research* 17: 699-705.

Harper, S.R. and S.R. Just. 1995. Design of Leachate Recirculation Systems for Municipal Solid Waste Landfills. Internal Draft of Unpublished Manuscript. August.

Harper, S.R., S.R. Just, C. Donohoe, and D.A. Carson. 1995. Design Elements Associated with Leachate Recirculation at Existing, Unlined Landfills. In *Seminar Publication on Landfill Bioreactor Design and Operation*. EPA/600/R-95/146. September.

Hartmann, K.H., and E. Hoffmann. 1990. Leachate treatment: Design recommendations for small but extremely fluctuating, highly polluted quantities of water. *Water Science and Technology* 22: 307-314.

Hartmans, S. and J. de Bont. 1992. Aerobic vinyl chloride metabolism in Mycobacterium arurum Li. *Applied and Environmental Microbiology* 58:1220.

Hartmans, S., J.A.M. de Bont, J. Tramper, and A.M. Luyben. 1985. Bacterial degradation of vinyl chloride. *Biotchnology Letters* 7(6):383-388.

Hater, G. R., 2001. Economics of Eight Scenarios for Landfill Bioreactors as Compared to a Base Case Subtitle D Landfill. *Waste Tech*. February 13, 2001.

Hauser, V.L. 1997. Agricultural engineering and the environmental field. ASAE paper no. 975072. Am. Soc. Of Agric. Engineers, 2950 Niles Rd., St. Joseph, MI 49085.

Hauser, V.L.; D.M. Gimon, D.E. Hadden, and B.L. Weand. 1999. *Survey of Air Force Landfills, Their Characteristics and Remediation Strategies*. Prepared for the Air Force Center for Environmental Excellence by Mitretek Systems. July.

Hauser, V.L. and M.A. Shaw. 1994a. Water movement through soil-vegetative landfill covers. ASAE paper no. 942176. Am. Soc. Of Agric. Engineers, 2950 Niles Rd., St. Joseph, MI 49085.

Hauser, V.L. and M.A. Shaw. 1994b. Climate effects on water movement through soil vegetative landfill covers. Seventeenth International Madison Waste Conference, September. Dept. of Engineering, Univ. of Wisconsin, Madison, WI.

Hauser, V.L., M.A. Shaw, and B.L. Weand. 1994. Effectiveness of soil-vegetative covers for waste sites. In *Proceedings, Superfund XV Conference*. Washington, DC. Hazardous Materials Control Resources Institute, Rockville, MD.

Hauser, V.L., M.A. Shaw, and B.L. Weand. 1995. *A Natural Cover for Buried Waste*. 21st Environmental Symposium and Exhibition, April 18-23. San Diego, CA.

Hauser, V.L., and B.L. Weand. 1998. Natural Landfill Covers. Third Tri-Service Environmental Technology Workshop, San Diego, CA. August.

Hauser, V.L., B.L. Weand, M.A. Shaw, and A.R. Wusterbarth. 1996. *Natural Covers for Landfills – A Closer Look*. 22st Environmental Symposium and Exhibition, March 18-21. Orlando, FL.

Hauser, V.L., Weand, B.L., and Gill, M.D. 2001a. Alternative Landfill Covers. Prepared for the Air Force Center for Environmental Excellence, Technology Transfer Division, Brooks AFB, TX. July.

Hauser, V.L., Weand, B.L., and Gill, M.D. 2001b. Natural Covers for Landfills and Buried Waste. Journal of Environmental Engineering, September, p. 768-775.

Hollinger, C., and W. Schumacher. 1994. *Antonie van Leeuwenhoek*. 66:239-246.

Hollinger, C., G. Schraa, A.J.M. Stams, and A.J.B. Zehnder. 1993. A highly purified enrichment culture couples the reductive dechlorination of tetrachloroethene to growth. *Applied and Environmental Microbiology* 59:2991-2997.

Hopkins, G.D., and P.L. McCarty. 1995. Field evaluation of in situ aerobic cometabolism of trichloroethylene and three dichloroethylene isomers using phenol and toluene as the primary substrates. *Environmental Science and Technology* 29:1628.

Hopkins, G.D., J. Munakata, and L. Semprini. 1993. Trichloroethylene concentration effects on pilot field-scale in-situ groundwater bioremediation by phenol-oxidizing microorganisms. *Environmental Science and Technology* 27:2542.

Hughes, J.B., K.L. Duston and C.H Ward. 2001. *Bioremediation*. Draft unpublished manuscript prepared for the Ground-Water Remediation Technologies Analysis Center (http://www.gwrtac.org).

Interstate Technology and Regulatory Cooperation Working Group (ITRC). 2000. *Presentation on Natural Attenuation of Chlorinated Solvents in Groundwater*. March.

IT Corporation. 2000. *Williams AFB, Mesa, AZ - LF-04*. Preliminary Draft. September.

Janseen, D.B., A. Scheper, L. Dijkhuizen, and B. Witholt. 1985. Degradation of halogenated aliphatic compounds by *Xanthobacter autotrophicus* GJ10. *Applied Environmental Microbiology* 49(3):673-677.

Javandel, I., and C.F. Tsang. 1986. Capture zone type curves: a tool for aquifer cleanup. *Ground Water* 24(5):616-625.

Johnson, C.A., M. Kaeppeli, S. Brandenberger, A. Ulrich, and W. Baumann. 1999. Hydrological and geochemical factors affecting leachate composition in municipal solid waste incinerator bottom ash. Part 2: The geochemistry of leachate from Landfill Lostorf, Switzerland. *Journal of Contaminant Hydrology* 40:239-259.

Johnson, C.A., G.A. Richner, T. Vitvar, T. Schittli, and M. Eberhard. 1998. Hydrological and geochemical factors affecting leachate composition in municipal solid waste incinerator bottom ash. Part 1: The hydrology of Landfill Lostorf, Switzerland. *Journal of Contaminant Hydrology* 33:361-376.

Johnston J.J., R.C. Borden, and M.A. Barlaz. 1996. Anaerobic Biodegradation of Alkylbenzenes and Trichloroethylene in Aquifer Sediments Down Gradient of a Sanitary Landfill. Journal of Contaminant Hydrology 23:263-283.

Kane, A.C., T.P. Wilson, and J.M. Fischer. 1997. Cometabolic biodegradation of trichloroethylene in microcosms. *In Situ and On-Site Bioremediation*, Volume 3. Battelle Press: Columbus, OH.

Koenigsberg, S.S., W.A. Farone, and C.A. Sandefur. 2000. Time-Release Electron Donor Technology for Accelerated Biological Reductive Dechlorination. In *Proceedings of the Second International Conference on Remediation of Chlorinated and Recalcitrant Compounds*, Monterey, CA. May 2000. Volume C2-4, p 39-46.

Kramer, F. 2000. Personal communication from Fran Kramer, USEPA, to Doug Downey of Parsons, August.

Kromann, A. and T.H. Christensen. 1998. Degradability of organic chemicals in a landfill environment studied by *in situ* and laboratory leachate reactors. Waste Manage Res 16(5): 437-445.

Lanzarone, N.A., and P.L. McCarty. 1990. Column studies on methanotrophic degradation of trichloroethene and 1,2-dichloroethane. *Ground Water* 28(6):910-919.

Lee, G.F., and A. Jones-Lee. 1997. *Deficiencies in USEPA Subtitle D Landfills in Protecting Groundwater Quality for as Long as MSW is a Threat: Recommended Alternative Approaches*. G. Fred Lee & Associates, El Macero, CA. March.

Lee, M.D., P.F. Mazierski, R.J. Buchanan, D.E. Ellis, and L.S. Sehayek. 1995. Intrinsic in situ anaerobic biodegradation of chlorinated solvents at industrial landfill. In *Intrinsic Bioremediation*. Battelle Press: Columbus, OH.

Lee, M.D., J.M. Odom, and R.J. Buchanan. 1998. New perspectives on microbial dehalogenation of chlorinated solvents: insights from the field. *Annual Review of Microbiology* 52:423-52.

Lesage, S., R.E.Jackson, M.W. Priddle, and P.G. Riemann. 1990. Occurrence and fate of organic solvent residues in anoxic groundwater at the Gloucester Landfill, Canada. *Environmental Science and Technology* 24(4):559-566.

Litherland, S.T., and D.W. Anderson. 1997. *In Situ and On-Site Bioremediation*, Volume 5. Battelle Press: Columbus, OH.

Ludvigsen, L., H.J. Albrechtsen, G. Heron, P.L. Bjerg, and T.H. Christensen. 1998. Anaerobic microbial redox processes in a landfill leachate contaminated aquifer (Denmark). *Journal of Contaminant Hydrology* 33:273-291.

Lyngkilde, J. and T.H. Christensen. 1992a. Redox zones of a landfill leachate pollution plume (Vejen, Denmark). *Journal of Contaminant Hydrology* 10: 273-289.

Lyngkilde, J. and T.H. Christensen. 1992b. Fate of organic contaminants in the redox zones of a landfill leachate pollution plume (Vejen, Denmark). *Journal of Contaminant Hydrology* 10: 291-307.

Magnuson, A. 1998. Leachate recirculation. *MSW Management* March/April.

Major, D.W., W.W. Hodgiuns, and B.J. Butler. 1991. Field and laboratory evidence of in situ biotransformation of tetrachloroethene to ethene and ethane at a chemical transfer facility in North Toronto. In *On Site Bioreclamation*. Hinchee, R.E., and R.F. Olfenbuttel, eds. San Diego, CA.

Maymo-Gatell, X., Y. Chien, J.M. Gossett, and S.H. Zinder. 1997. Isolation of a bacterium that reductively dechlorinates tetrachloroethene to ethene. *Science* 276:1568-1571.

McCarty, P.L. 1997a. Biotic and abiotic transformations of chlorinated solvents in ground water. In *EPA Proceedings of The Symposium in Natural Attenuation of Chlorinated Organics in Ground Water*. Dallas, TX. EPA/540/R-97/504/1997. September 11-13, 1996.

McCarty, P.L. 1997b. Breathing with chlorinated solvents. *Science* 276:1521.

McCarty, P.L., and L. Semprini. 1994. Ground-water treatment for chlorinated solvents. In *Handbook of Bioremediation*. Lewis Publishers: Boca Raton, FL.

McCarty, P.L., M.N. Goltz, G.D. Hopkins, M.E. Dolan, J.P. Allan, B.T. Kawakami, and T.J. Carrothers. 1998. Full-scale evaluation of in situ cometabolic degradation of trichloroethylene in groundwater through toluene injection. *Environmental Science and Technology* 32:88-100.

McCray, J.E., R. Statom, and G.D. Thyne. 2001. *Types of Landfill Caps that will Promote Natural Attenuation of Subsurface CAH Contaminants via Physical and Geochemical Controls, and Influence of Groundwater Recirculation on Natural Attenuation in Landfills*. Prepared by Colorado School of Mines for Parsons. Denver, CO. May 12.

McCray, J.E., G.D. Thyne, and R. Statom. 2001. *Final Literature Review: Natural Attenuation of CAH Compounds in Landfills: Influence of Hydrology, Geochemistry, and Groundwater Recirculation*. Prepared by Colorado School of Mines for Parsons. Denver, CO. June 7.

McCreanor, P.T, and D.R. Reinhart. 2000. Mathematic modeling of leachate routing in a leachate recirculating landfill. *Water Resources* 34(4):1285-1295.

McDonald and Harbaugh. 1988. A modular three-dimensional finite-difference groundwater flow model. *US Geological Survey Techniques of Water Resources Investigations*, Book 6, Chapter A1.

Means. 2001. Environmental Remediation Cost Data-Unit Price. 7th Annual Edition. R.S. Means Company and Talisman Partners, Ltd.

Miller, D.E., and S.M. Emge. 1997. Enhancing landfill leachate recirculation system performance. *Practice Periodical of Hazardous, Toxic, and Radioactive Waste Management* 1(3):113-119.

Mirecki, J.E. and W.S. Parks. 1994. Leachate Geochemistry at a Municipal Landfill in Memphis, Tennessee. Ground Water 32(3): 390-398.

Mravik, S.C., G.W. Sewell and A.L. Wood. 2000. Co-Solvent-Based Source Remediation Approaches. In: Proceedings of the Second International Conference on Remediation of Chlorinated and Recalcitrant Compounds, Monterey, CA. Volume C2-5 (Physical and Thermal Treatment Technologies), p 269-276. Battelle Press.

Munakata-Marr, J., V.G.Matheson, L.J. Forney, J.M. Tiedje, and P.L. McCarty. 1997. Long-term biodegradation of trichloroethylene influenced by bioaugmentation and dissolved oxygen in aquifer microcosms. *Environmental Science and Technology* 31:786-791.

NAVFACHQ, 2002. Project correspondence from Martha Midgette of NAVFACHQ. Data from NORM database.

Neumann, A., H. Scholz-Muramatsu, and G. Diekert. 1994. Tetrachloroethene metabolism of *Dehalospirillum* multivorans. Archive of *Microbiology* 162:295-301.

Newell, C., P. Haas, J. Hughs, and T. Khan. 2000. Results from Two Direct Hydrogen Delivery Field Tests for Enhanced Dechlorination. In *Proceedings of the Second International Conference on Remediation of Chlorinated and Recalcitrant Compounds*, Monterey, CA. May 2000. Volume C2-4, p 31-37.

Nielsen, D.M. 1991. *Practical Handbook of Ground-Water Monitoring*. Lewis Publishers: Chelsea, MI.

Nielsen, P.H., H. Bjarnadottir, P.L. Winter, and T.H. Christensen. 1995. In situ and laboratory studies on the fate of specific organic compounds in an anaerobic landfill leachate plume, 2. Fate of aromatic and chlorinated aliphatic compounds. *Journal of Contaminant Hydrology* 20: 51-66.

Nyhan, J.W., T.E. Hakonson, and A. Drennon. 1990. A water balance study of two landfill cover designs for semiarid regions. Journal of Environmental Quality 19(2):281-288.

Oman, C., and H. Rosqvist. 1999. Transport fate of organic compounds with water through landfills. *Water Resources* 33(10):2247-2254.

Onay, T.T., and F.G. Pohland. 1998. In situ nitrogen management in controlled bioreactor landfill. *Water Research* 32(5):1383-1392.

Pacey, J., D. Augenstein, R. Morck, D. Reinhart, and R. Yazdani. 1999. *The Bioreactive Landfill*. MSW Management (Online Publication: via Forester Communications, Inc.).

Pankow, J.F., and J.A. Cherry. 1996. *Dense Chlorinated Solvents and Other DNAPLs in Groundwater*. Waterloo Press, Portland, Oregon.

Parsons Engineering Science, Inc. 1999. Natural Attenuation of Chlorinated Solvents, Performance and Cost Results from Multiple Air Force Demonstration Sites, Technology Demonstration Technical Summary Report. Prepared for the Air Force Center for Environmental Excellence, Technology Transfer Division, Brooks AFB, Texas. Denver, Colorado. October.

Patel, R.N., C.T. Hou, A.I. Laskin, and A. Felix. 1982. Microbial oxidation of hydrocarbons: properties of a soluble methane monooxygenase from a facultative methane-utilizing organism, *Methylobacterium* sp. Strain CRL-26. *Applied and Environmental Microbiology* 44:1130-1137.

Pohland, F.G. 1980. Leachate recycle as landfill management option. *Journal of Environmental Engineering* 106:1057-1069.

Pohland, F.G., and S.R. Harper. 1985. *Critical Review and Summary of Leachate and Gas Production from Landfills*. US Environmental Protection Agency, EPA/600/2-86/073.

Pohland, F.G., W.H. Cross, J.P. Gould, and D.R. Reinhart. 1993. *Behavior and Assimilation of Organic and Inorganic Priority Pollutants Codisposed with Municipal Refuse*. National Techinal Information Service: Springfield, VA. PB 93-227198A5.

Pohland, F.G., F. Karadagli, J.C. Kim, and F.P. Battaglia. 1998. Landfill codisposal of pentachlorophenol (PCP)-treated waste wood with municipal solid waste. *Water Science and Technology* 38(2):169-175.

Pohland, F.G., and J.C. Kim. 2000. Microbially mediated attenuation potential of landfill bioreactor systems. *Water Science and Technology* 41(3):247-254.

Pohland, F.G., A.B. Al-Yousfi, and D.R. Reinhart. 2001. Anaerobic digestion of organic solid waste in bioreactor landfills. In *Biomethanization of the Organic Fraction of Municipal Solid Wastes*. J. Mata-Alvarez, ed. IWA Publishing. In press.

Pohland, F.G., and J.C. Kim. 2001. *Transformation of Chlorinated Aliphatic Hydrocarbons Associated with Landfill Disposal Practices.* Prepared by University of Pittsburgh for Parsons. April 9.

Reineke, W., and H.J. Knackmuss. 1984. Microbial metabolism of haolaromatics: isolation and properties of a chlorobenzene-degrading bacterium. *European Journal of Applied Microbiology and Biotechnology* 47:395-402.

Reinhard, M., G.P. Curtis, and M.R. Kriegman. 1990. *Abiotic Reductive Dechlorination of Carbon Tetrachloride and Hexachloroethane by Environmental Reductants: Project Summary*. EPA/600/S2-90/040. September.

Reinhard, M., N.L. Goodman, and J.F. Barker. 1994. Occurrence and distribution of organic chemicals in two landfill leachate plumes. *Environmental Science and Technology* 18(12):953-961.

Reinhart, D.R., and F.G. Pohland. 1991. The assimilation of organic hazardous wastes by municipal solid waste landfills. *Journal of Industrial Microbiology* 8:193-200.

Reinhart, D.R., and T. Townsend. 1998. *Landfill Bioreactor Design and Operation.* CRC Press LLC: Boca Raton, FL.

Reinhard, D.R., P.T. McCreanor, and T. Townsend. 2001. *The Bioreactor Landfill: Its Status and Future.* Waste Management and Research Journal. Anticipated Publication Date: April.

Remediation Technologies Development Forum (RTDF). 1997. *Natural Attenuation of Chlorinated Solvents in Groundwater: Principles and Practices*. Version 3.0. August.

Sander, P., R.M. Wittaich, P. Fortnagel, H. Wilkes, and W. Francke. 1991. Degradation of 1,2,4-trichloro- and 1,2,4,5-tetrachlorobenzene by *Pseudomonas* strains. *Applied Environmental Microbiology* 57:1430-1440.

Sanin, F.D., R.U. Knappe, and M.A. Barlaz. 2000. The fate of toluene, acetone, and 1,2-dichlorethane in a laboratory-scale simulated landfill. *Water Resources* 34(12):364-3074.

Satkin, R.L., and P.B. Bedient. 1988. Effectiveness of various aquifer restoration schemes under variable hydrogeologic conditions. *Ground Water* 26(4):488-498.

Shafer, J.M, 1987. Reverse pathline calculation of time-related capture zones in nonuniform flow, *Ground Water, 25*(3), 283-289.

Scholz-Muramatsu. 1995. Isolation and characterization of *Dehalospirillum multivorans* gen sp. Nov., a tetrachloroethene-utilizing, strictly anaerobic bacterium. *Archive of Microbiology* 163:48-56.

Schraa, G., M.L. Boone, M.S.M. Jetten, A.R.W. van Neerven, P.J. Colberg, and A.J.B. Zehnder. 1986. Degradation of 1,2-dichlorobenzene by *Alcaligene sp.* Strain A175. *Applied Environmental Microbiology* 52:1374-1381.

Schroeder, P.R., C.M. Lloyd, and P.A. Zappo. 1994. The Hydrologic Evaluation of Landfill Performance (HELP) Model, User's Guide for Version 3. EPA/600/R-94/168a.

Schultz, B. and P. Kjeldsen. 1986. Screening of organic matter in leachate from sanitary landfills using gas chromatography combined with mass spectrometry. Water Res. 20: 965.

Semprini, L. 1995. Anaerobic transformation of chlorinated aliphatic hydrocarbons in a sand aquifer based on spatial chemical distributions. *Water Resources Research* 31(4):1051-1062.

Semprini, L., P.V. Roberts, and G.D. Hopkins. 1990. A field evaluation of in-situ biodegradation of chlorinated ethenes: results of biostimulation and biotransformation experiments. *Ground Water* 28:715-727.

Sewell, G.W. 1998. Designing and applying treatment technologies. *Remediation of Chlorinated and Recalcitrant Compound*s. Battelle Press. Columbus, Ohio. p 9-14.

Sharma, P.K., and P.L. McCarty. 1996. Isolation and characterization of a facultatively aerobic bacterium that reductively dehalogenates tetrachloroethene to cis-1,2-dichloroethene. *Applied Environmental Microbiology* 62:761-765.

Smatlak, C.R., J.M. and Gossett. 1996. Comparative kinetics of hydrogen utilization for reductive dechlorination of tetrachloroethene and methanogenesis in an anaerobic enrichment culture. *Environmental Science and Technology* 30:2850-2858.

Solid Waste Association of North America (SWANA). 2000. Comments on Docket N. F-1999-MLFN-FFFFF, 610 Review of the Solid Waste Disposal Facility Criteria (40 CFR Part 258). Submitted to the RCRA Information Center, United States Environmental Protection Agency. February.

Spuij, F.A., A. Alphenaar, H. de Wit, R. Lubbers, K. Brink, J. Gerritse, J. Gottschal, and S. Houtman. 1997. *In Situ and On Site Bioremediation,* Volume 5. Battelle Press: Columbus, OH.

S.S. Papadopulous & Associates. 1996. MT3D^{99}. December.

Stucki, G., U. Krebser, and T. Leisinger. 1983. Bacterial growth on 1,2-dichloroethane. *Experentia* 39:1271-1273.

Suflita, J.M., C.P. Gerba, R.K. Ham, A.C. Palmisano, and J.A. Robinson. 1992. The world's largest landfill. *Environmental Science & Technology* 26(8):1486-1495.

Suthersan, S. 2001. Natural and Enhanced Remediation Systems. Lewis Publishers. 419 pg.

Tandol, V. 1994. Reductive dehalogenation of chlorinated ethenes and halogenated ethanes by a high-rate anaerobic enrichment culture. *Environmental Science and Technology* 28:973-979.

Thornton, S.F., M.I. Bright, D.N. Lerner, and J.H. Tellam. 2000. Attenuation of landfill leachate by UK Triassic sandstone aquifer materials 2: sorption and degradation of organic pollutants in laboratory columns. *Journal of Contaminant Hydrology* 43:355-383.

Thornton, S.F., J.H. Tellam, and D.N. Lerner. 2000. Attenuation of landfill leachate by UK Triassic sandstone aquifer materials 1: fate of inorganic pollutants in laboratory column. *Journal of Contaminant Hydrology* 43:327-354.

Tittlebaum. 1982. Organic carbon content stabilization through landfill leachate recirculation. Journal of the Water Pollution Control Federation 34 (5): 428-433.

Tovanabootr, A., S. Russel, N.H. Stoffers, D.J. Arp, and L. Semprini. 1997. An evaluation of five aerobic cometabolic substrates for trichloroethylene treatment by microbes stimulated from the subsurface of McClellan Air Force Base. *In Situ and On-Site Bioremediation*, Volume 3. Battelle Press: Columbus, OH.

Townsend, T.G., W.L. Miller, and J.F.K. Earle. 1995. Leachate-recycle infiltration ponds. *Journal of Environmental Engineering* 121(6):465-471.

Townsend, T.G., W.L. Miller, H.J. Lee, J.F.K. Earle. 1996. Acceleration of landfill stabilization using leachate recycle. *Journal of Environmental Engineering* 122(4):263-268.

Trabelsi, I., H. Horibe, N. Tanaka, and T. Matsuto. 2000. Origin of low carbon/nitrogen ratios in leachate from old municipal solid waste landfills. Waste Manage Res 18: 224-234.

Travis, B.J., and N.D. Rosenberg. 1997. Modeling in situ bioremediation of TCE at Savannah River: Effects of product toxicity and microbial interactions on TCE degradation. *Environmental Science and Technology* 31:3093-3102.

Turpie, A., Lizotte, C., DeFlaun, M., Quinnan, J., and Marley, M. 2000. Performance of Field-Scale Sequential Anaerobic-Aerobic Bioremediation Demonstration. In *Proceedings of the Second International Conference on Remediation of Chlorinated and Recalcitrant Compounds*, Monterey, CA. May. Volume C2-4, pg. 319-326. Battelle Press.

United States Air Force (USAF). 1997. *Presumptive Remedy Engineering Evaluation/Cost Analysis (PREECA), Final*. Second edition. Prepared by Radian Corporation.

USAF. 1998. *Installation Restoration Program, Aerobic Cometabolic In Situ Bioremediation Technology Guidance Manual and Screening Software User's Guide*. Center for Environmental Excellence, Environmental Restoration Division. Brooks Air Force Base, TX. June. http://en.afit.af.mil/env/insitubio.htm .

United States Army Corps of Engineers (USACE), 2001. Project correspondence with Dave Becker - US Army Corps of Engineers, Omaha District and Rob Snyder - US Army Headquarters.

United States Environmental Protection Agency (USEPA). 1985. Remedial Action at Waste Disposal Sites. EPA/625/6-85/006.

USEPA. 1988. *Guide to Technical Resources for the Design of Land Disposal Facilities*. Risk Reduction Research Laboratory, US Environmental Protection Agency, Cincinnati, OH. EPA/625/6-88/018.

USEPA. 1991. *Conducting Remedial Investigations/Feasibility Studies for CERCLA Municipal Landfill Sites.* Office of Emergency and Remedial Response, OSWER Directive 9355.3-11, EPA 540/P-91/001. February.

USEPA. 1993a. *Criteria for Solid Waste Disposal Facilities: A Guide for Owners/Operators.* Office of Solid Waste and Emergency Response. EPA 530-SW-91-089. March.

USEPA. 1993b. *Presumptive Remedy for CERCLA Municipal Landfill Sites.* Office of Solid Waste and Emergency Response, Directive No. 9355.0-49FS. EPA 540-F-93-035. September.

USEPA. 1995. *Presumptive Remedies: CERCLA Landfill Caps RI/FS Data Collection Guide.* Solid Waste and Emergency Response, Directive 9355.3-18FS. EPA/540/F-95/009. August.

USEPA. 1996. *Application of the CERCLA Municipal Landfill Presumptive Remedy to Military Landfills.* Office of Solid Waste and Emergency Response, Directive No. 9355.0-67FS. EPA/540/F-96/020. December.

USEPA. 1997a. *Landfill Presumptive Remedy Saves Time and Cost.* Office of Solid Waste and Emergency Response, Directive No. 9355.0-661. EPA 540/F-9/017. January.

USEPA. 1997b. *Best Management Practices (BMPs) for Soils Treatment Technologies: Suggested Operational Guidelines to Prevent Cross-Media Transfer of Contaminants During Cleanup Activities.* Office of Solid Waste and Emergency Response. EPA 530-R-97-007. May.

USEPA. 1998a. *RCRA, Superfund & EPCRA Hotline Training Module: Introduction to Applicable or Relevant and Appropriate Requirements (ARARs).* Office of Solid Waste and Emergency Response. EPA 540-R-98-020. June.

USEPA. 1998b. *Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Ground Water*. Office of Research and Development. Washington, D.C. EPA/600/R-98/128. September.

USEPA. 1999a. *Summary of the Remediation Technologies Development Forum.* Phytoremediation Action Team Alternative Cover Assessment Program (ACAP) Meeting, March 22 and 23. San Francisco, CA.

USEPA. 1999b. *Use of Monitored Natural Attenuation at Superfund, RCRA, Corrective Action, and Underground Storage Tank Sites*, Office of Solid Waste and Emergency Response, Directive No. 9200.4-17P. April.

USEPA. 1999c. *Yolo County Bioreactor Landfill XL Project Proposal*. Submitted by the Planning and Public Works Department of Yolo County and Ramin Yazdani, Chief County Engineer. September.

USEPA. 1999d. *Landfill Reuse: Notes on an Engineering Forum Roundtable Discussion*. Prepared by Chet Janowski, Robert Stamnes, Frank Vavra, JoAnn Cola, and Camille Hueni on the USEPA Technical Support Project Meeting, Orleans Hotel, Las Vegas, NV. October 28.

USEPA. 1999e. *Groundwater Issues*. Office of Research and Development. EPA/540/S-99/001. September.

USEPA. 2000a. *EPA Project XL: Final Project Agreement for the Yolo County Accelerated Anaerobic and Aerobic Composting (Bioreactor) Project*. Submitted by the Planning and Public Works Department of Yolo County and Ramin Yazdani, Chief County Engineer. February.

USEPA. 2000b. Alternative liner performance, leachate recirculation, and bioreactor landfills: request for information and data. 65 *Federal Register* 18014. April 6.

USEPA. 2000c. *Environmental Fact Sheet: EPA Requests Information on Bioreactor Landfills and Performance of Alternative Liners for Landfills when Leachate is Recirculated*. Office of Solid Waste and Emergency Response, EPA530-F-00-023. April.

USEPA. 2000d. *USEPA Technical Support Project Semi-Annual Meeting Technical Session Minutes*. Washington, DC. April 25-28.

USEPA. 2000e. *Engineered Approaches to In Situ Bioremediation of Chlorinated Solvents: Fundamentals and Field Applications*. Office of Solid Waste and Emergency Response. EPA542-R-00-008. July.

USEPA. 2000f. *Results-Based Approaches to Corrective Action: Overview*. Office of Solid Waste. July.

USEPA. 2000g. *Memorandum on the Applicability of RCRA Section 3020 to In-Situ Treatment of Groundwater*. Prepared by Elizabeth Cotsworth, Director, Office of Solid Waste. December 27.

USEPA. 2001a. Summary of number of MSW landfills in U.S. www.epa.gov. September.

USEPA. 2001b. *RCRA Cleanup Reforms: Reforms II, Fostering Creative Solutions*. Office of Solid Waste and Emergency Response. EPA 530-F-01-001. January.

USEPA. 2001c. Project XL site-specific rulemaking for Yolo County Landfill, Davis, Yolo County, California (final rule). 66 *Federal Register* 42441. August 13.

USEPA. 2001d. Project XL site-specific rulemaking for Buncombe County Landfill, Alexander, Buncombe County, North Carolina (final rule). 66 *Federal Register* 44061. August 22.

USEPA. 2001e. *Use of Bioremediation at Superfund Sites.* Solid Waste and Emergency Response. EPA 542-R-01-019. September.

Vogel, T.M., and P.L. McCarty. 1987a. Abiotic and biotic transformation of 1,1,1-trichloroethane under methanogenic conditions. *Environmental Science and Technology* 21:1208-1213.

Vogel, T.M., and P.L. McCarty. 1987b. Rate of abiotic formation of 1,1-dichloroethylene from 1,1,1-trichloroethane in groundwater. *Journal of Contaminant Hydrology* 1:299-308.

Vogel, T.M., C.S. Criddle, and P.L. McCarty. 1987. Transformations of halogenated aliphatic compounds. *Environmental Science and Technology* 22:722-736.

Warren, R.W., T.E. Hakonson, and K.V. Bostick. 1996a. The hydrologic evaluation of four cover designs for hazardous waste landfills at Hill Air Force Base. *Federal Facilities Environmental Journal* 6(1):91-110.

Warren, R.W., T.E. Hakonson, and K.V. Bostick. 1996b. Choosing the most effective hazardous waste landfill cover. *Remediation*, Spring, pp. 23-41.

Weand, B.L., and V.L. Hauser. 1997. The evapotranspiration cover. *Environmental Protection* 8(11):40-42.

Weand, B.L., J.D. Horin, V.L. Hauser, D.M. Gimon, M.D. Gill, M. Mehta, and D.J. Casagrande. 1999. *Landfill Covers for Use at Air Force Installations*. Prepared for Air Force Center for Environmental Excellence, Technology Transfer Division. February.

Wiedemeier, T.H., H.S. Rifai, C.J. Newell, and J.T. Wilson. 1999. *Natural Attenuation of Fuels and Chlorinated Solvents in the Subsurface*. John Wiley & Sons, Inc.: Canada.

Wilson, J.T., and B.H. Wilson. 1985. Biotransformation of trichloroethylene in soil. *Applied Environmental Microbiology* 29:242-243.

Wong, K.F., V.G. He, and H. Solo-Gabiele. 1997. *Infiltration Rates Through Synthetic Caps and Slide Slopes at Landfills*. Report 97-12, Palm Beach Solid Waste Authority. Palm Beach, FL.

Workman, D.J., S.L. Woods, Y.A. Gorby, J.K. Fredrickson and M.J. Truex. 1997. Microbial reduction of Vitamin B12 by *Shewanella* alga Strain BrY with subsequent transformation of carbon tetrachloride. *Environmental Science and Technology* 31:2292-2297.

Yager, R.M., S.E. Bilotta, C.L. Mann, and E.L. Madsen. 1997. Metabolic and in situ attenuation of chlorinated ethenes by naturally occurring microorganisms in a

fractured dolomite aquifer near Niagara Falls, New York. *Environmental Science and Technology* 31:3138-3147.

Yagi, M. 1994. Bioremediation of trichloroethylene-contaminated soils by a methane-utilizing bacterium Methylocystis. In *Bioremediation of Chlorinated and Polycyclic Aromatic Hydrocarbon Compounds*. Lewis Publishers: Boca Raton, FL.

Young, J.D., D.J. Altman, K.H. Lombard, A.W. Bourquin, D.C. Mosteller, and T.C. Hazen. 1997. Sanitary landfill optimization test for remediation of chlorinated solvents. *In Situ and On-Site Bioremediation*, Volume 3. Battelle Press: Columbus, OH.

APPENDIX A

POINTS OF CONTACT

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IMPACT OF LANDFILL CLOSURE DESIGNS ON LONG-TERM NATURAL ATTENUATION OF CHLORINATED HYDROCARBONS
ESTCP CONTRACT # DACA72-00-C-0013

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APPENDIX B

SITE-SPECIFIC GROUNDWATER CHEMISTRY DATA

Table B.1
Air Force Base Landfill Chemistry Summary
Military Landfill Closure Design Evaluation

Installation and Well IDs	Water Level Elevation (feet amsl[a/])	Well Location Relative to Landfill	PCE	TCE	cis 1, 2-DCE	VC	Methane	Ethene	Ethane	TOC	Dissolved Oxygen	Nitrate + Nitrite	Ferrous Iron	Sulfide	Sulfate	Alkalinity	CO_2	Ammonia	Chloride	COD	Redox Potential	pH
			(µg/L[b/])				(mg/L[c/])														(mV[d/])	
Williams AFB (Arizona) LF-04																						
LF01-W13	1188.34	upgradient	ND [e/]	ND	ND	ND	[f/]															
LF01-W07	1179.15	within	2.5	0.33J [g/]	ND	ND																
LF01-W09A	1173.08	downgradient	4.8	1.1	ND	ND																
LF01-W17	1162.88	downgradient	0.35J	0.37J	ND	ND																
Griffiss AFB (New York) LF 1																						
LF1MW-1	531.45	upgradient	ND	ND	ND	ND				2.5	ND	0.35F			13.2	85		ND	6.57	9.26	64.6	6.69
LF1MW101	505	adjacent/downgradient	ND	0.34F [g/]	0.58F	4.45				6.84	0.9	ND			0.57F	440		3.57	2.82F	21.43	-29.5	6.84
LF1P2	498.81	downgradient	ND	ND	ND	0.25F				3.29	0.6	ND			0.04F	438		1.87	16	17.8	-37.2	7.23
LF1MW-5	493	downgradient	ND	0.27F	0.59F	0.77F				3.07	0.11	ND			0.46F	498		2.72	46	28.9	-26.5	6.39
LF1P-5	491.36	downgradient	ND	ND	ND	ND				1.48	0.73	ND			0.09F	310		0.43	ND	ND	8.43	7.07
Griffiss AFB (New York) LF 7																						
LF7MW-16	502.81	adjacent/upgradient	ND	0.32F	ND					24	0.09	ND			48.1	592		ND	4.03F	53.22	-6.5	6.45
LF7MW-17	491.76	adjacent/downgradient	ND	ND	0.33F					6.87	2.35	2.83F			81.8	504		ND	13.4	11.57	261.5	7.24
LF7MW-22	483	downgradient	ND	5.25	0.69F					3.43	0.72	ND			102	560		ND	13.2	6.94	-46.6	6.5
Westover AFB (Massachusetts) Landfill B CAAA																						
B-4	243.14	upgradient	ND	ND		ND	0.65			0.6	9.36	0.51	0		53				10.2		118.4	NA
GP-503	241.47	within	ND	ND		ND	1998			15.6	5.52	0.09	52		96.6				2.6			NA
GP-509	238.08	within	ND	0.6J		90E [g/]	5280			15	0.24	0.225	32.2		58.9				6.9		-4.9	5
PES-6D	236.52	downgradient	ND	ND		14	321			6.2	3.99	0.095	24.5		0				18.9		9.28	12.78
FE Warren AFB (Wyoming) LF03																						
196	6129.97	upgradient	ND	ND	ND	0.6	ND	ND	ND	1.93		1.65	ND	ND	16.2	120	15	ND	2.54			
208	6118.89	adjacent	ND	20	16	ND	ND	ND	ND	7.93	1.4	19.5	ND	ND	206	240	35	ND	66.6		182	6.83
62	6118.61	downgradient	ND	20	12.2	ND	ND	ND	ND	3.34	2.65	7.65	ND	ND	47.2	160	25	ND	20.8		177	7.45
147	6115.7	downgradient	ND	15	8.2	ND	ND	ND	ND	5.45	3.8	9.17	ND	ND	55.4	200	35	ND	31.4		102	
FE Warren AFB (Wyoming) LF5a																						
LF5-4	6187.18	within/upgradient	`	ND	ND					1.85J		8.45			43.7			ND	38.7			
MW246	6169.94	downgradient		ND	ND					ND		3.4			15.5				8.58			
FE Warren AFB (Wyoming) LF6																						
MW-202B	6167.31	upgradient	ND	ND	ND	ND					8.05		0.02			132					218.8	7.47
MW-29	6155	within	0.98	0.77	ND	ND					2.15		0.1			240					74.8	6.61
MW-337	6150	downgradient	ND	ND	ND	ND					4.87		0.48			115					-64.4	7.48
Kelly AFB (Texas)																						
LF014PW046	628.93	upgradient					ND	ND	ND	ND	3.8	5.04	ND	ND	20	280	15	ND	13.4		587	
LF014MW082	622.55	adjacent					ND	ND	ND	1.58	3.15	4.66	ND	ND	ND	320	15	ND	12.9		486	
LF014TW102	620.02	within	ND	0.39J	77.5	105	0.3	ND	ND	4.01	0.2	ND	4.11			480	70		69.5		-76	6.7
LF014TW106	622.29	within	ND	0.12J	5.39	339	0.08	ND	ND	2.71	0.22	1.66	3.4	ND	48	400	32		34.6		-135	6.4
LF014TW108	622.48	within	ND	ND	203	85.9	0.26	ND	ND	2.93	0.17	2.05	0.39	ND	43	400	26		45.2		-51	7.0
LF014TW116	622.58	within	ND	43.9	225	5.36	0.06	ND	ND	3.7	0.04	2.22	3.06	ND	4	340	26		16.8		-105	7.0
Plattsburgh AFB (New York) LF-023																						
MW-23-002	219.74	upgradient	ND	ND	ND	ND				2.4		1.5			37.8	100		NA	301	10		
MW-23-006	215.94	adjacent	ND	ND	ND	ND				2.2		3.1			37.5	98		NA	339	ND		
MW-23-011	212.37	downgradient	ND	ND	ND	ND				3.5		0.1			24.7	170		NA	158	20		

[a/] amsl = above mean sea level
[b/] µg/L = micrograms per liter
[c/] mg/L = milligrams per liter
[d/] mV = millivolts
[e/] ND = not detected
[f/] shading indicates parameter not measured
[g/] J = estimated concentration; F = analyte concentration is between the method detection limit and the reporting limit; E = result exceeds instrument calibration range
PCE = tetrachloroethene, TCE = trichloroethene, DCE = dichloroethene, VC = vinyl chloride, TOC = total organic carbon, CO_2 = carbon dioxide, COD = chemical oxygen demand, redox potential = reduction-oxidation potential