HUMAN HEALTH AND COASTAL ECOSYSTEM RISK ASSESSMENT OF THE MASSACHUSETTS MILITARY RESERVATION MAIN BASE LANDFILL GROUNDWATER PLUME

by

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Human Health And Coastal Ecosystem Risk Assessment Of The Massachusetts Military Reservation Main Base Landfill Groundwater Plume

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Submitted to the Department of Civil and Environmental Engineering on May 24, 1996, in Partial Fulfillment of the Requirements for the Degree of Master of Engineering in Civil and Environmental Engineering

Abstract

A plume of contaminated groundwater emanating from the main base landfill of the Massachusetts Military Reservation is predicted to discharge into Red Brook and Megansett harbors of Buzzards Bay, MA. This contaminant plume contains metals above the background levels for the MMR and volatile organic chemicals. The potential discharge of these contaminants especially metals into the shallow coastal harbors poses a threat to the shellfish population of these areas as well as to consumers of shellfish.

An assessment of the risk posed to public health from consumption of quahog clams contaminated with metals from the landfill groundwater plume was performed using a standard United States Environmental Protection Agency (USEPA) risk assessment methodology. By applying the maximum detected concentrations of metals in the plume, the maximum total cancer and non-cancer risks to lifetime consumers of tainted quahogs from Red Brook and Megansett harbors were found to be above the acceptable USEPA standards. These calculated risks are conservative estimates since worst case assumptions were utilized in the assessment.

A qualitative estimate of the risks posed to the coastal ecosystem from the potential bioaccumulation of metals in quahog clams predicted that the quahog population will decline due to the accumulation of metals in their body tissues. The decrease in quahog populations is expected to lead to a reduction in the populations of other marine species that depend on quahogs as their primary food source. The potential transfer of bioaccumulated metals in quahogs to higher order organisms will also cause deleterious effects to the ecosystem.

Thesis Supervisor: David H. Marks
Title: Professor of Civil and Environmental Engineering

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Chapter 1 Introduction

The Massachusetts Military Reservation (MMR), a military base located in Cape Cod, MA, has conducted military training exercises at the reservation since 1935. Due to the activities of the various military units that have occupied the base since its establishment, large areas of the groundwater resource of western Cape Cod has been contaminated with heavy metals and organic chemicals. Figure 1 shows the planview extent of groundwater contamination emanating from the MMR. The magnitude of the contamination at the MMR prompted the United States Environmental Protection Agency (USEPA) in 1989 to place the reservation on the National Priorities List (NPL) of Superfund sites. Sites on the NPL are to be remediated prior to other Superfund sites.

The focus of the current study is the contamination plume originating from the main base landfill of the MMR. This plume, one of the largest at the MMR site, is termed the LF-1 plume. The aerial extent of the LF-1 plume contamination is delineated in Figure 1. The constituents of the groundwater pollution from the landfill consist mainly of volatile organic compounds and heavy metals. A number of these chemicals have been identified as human carcinogens. The LF-1 plume is generally migrating westward from the landfill and within 2 to 2.5 years, is anticipated to reach the water supply wells for the town of Bourne. The contaminants are most likely to discharge into Red Brook and Megansett harbors of Buzzards Bay, MA, if the plume is permitted to run its course. The flow of tainted groundwater from the military reservation toward Buzzards Bay poses potential

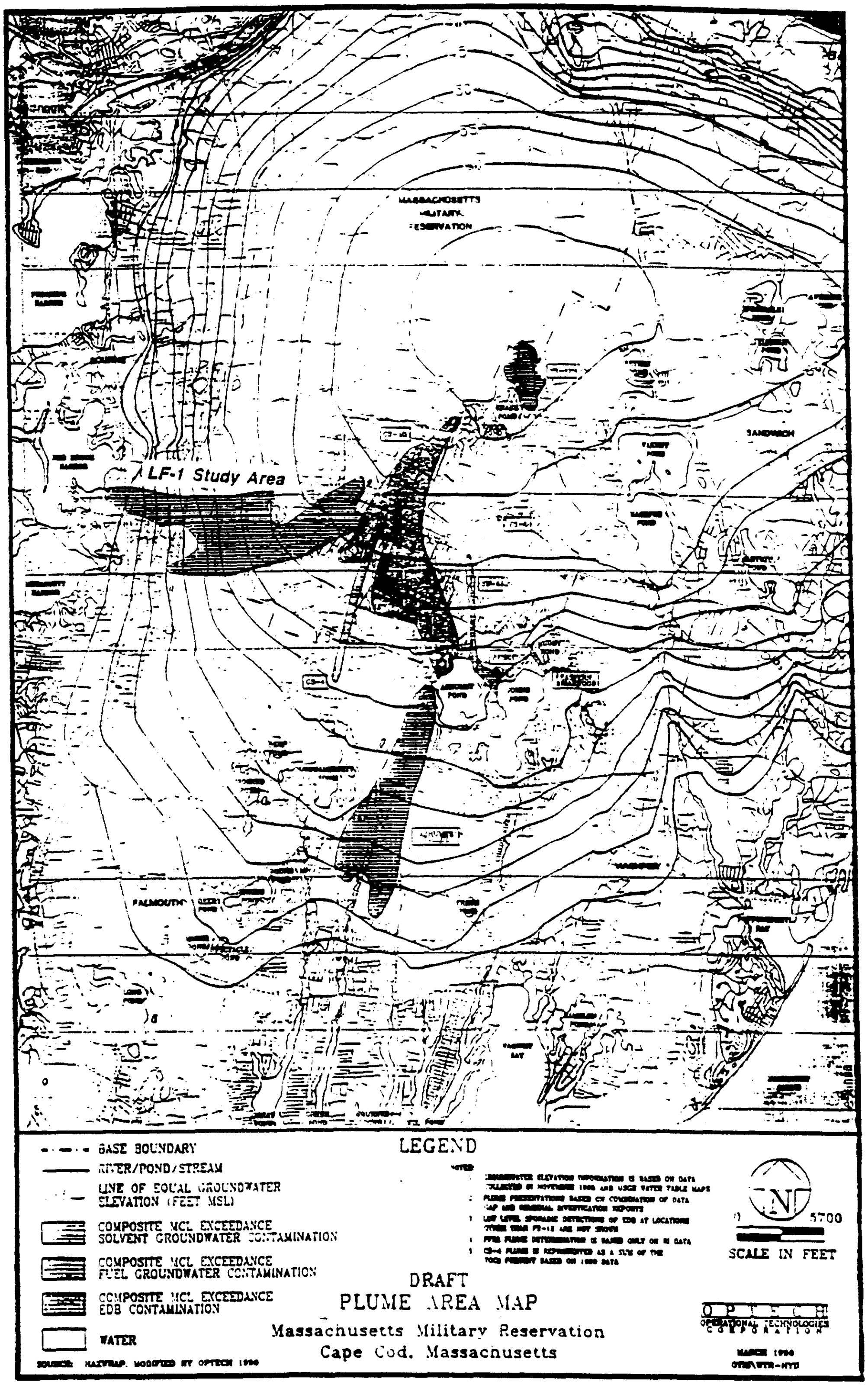

Figure 1 Groundwater contamination emanating from the MMR

human health and ecological risks for the residents of the towns of Bourne and Falmouth in western Cape Cod.

CDM Federal (1995) in the *Remedial Investigation: main base landfill and hydrogeologic region I study* performed a preliminary risk assessment for the LF-1 plume. The maximum cancer risk found for adult residents of the towns of Bourne and Falmouth for future exposure to contaminated groundwater is 1.3E-03. This risk is interpreted as the incremental increase in probability of developing cancer above the background level for each exposed resident. The USEPA acceptable cancer risk standard ranges from 1.0E-06 to 1.0E-04. The standard is set independently for each site and case. The increased risk of 1.3E-03 for each resident is above the highest acceptable USEPA standard. In addition, the overall maximum Hazard Index (HI) for non-cancer risk from potential exposure to the contaminated groundwater is 39.5. The USEPA's acceptable HI standard for non-cancer risk is 1.0. Calculated HI that are above the USEPA standard pose possible non-cancer deleterious health effects to exposed populations. Thus, the LF-1 plume delineated by CDM Federal poses cancer and non-cancer risks to adult residents of Bourne and Falmouth above the USEPA acceptable standard.

Operational Technologies Corporation (OpTech, 1996) proposed a pump and treat system to contain the majority of the LF-1 plume and to reduce the future potential risk to humans and the coastal ecosystem. The containment strategy is a row of extraction wells along Route 28 of western Cape Cod. Figure 2 shows the locations of the proposed extraction wells.

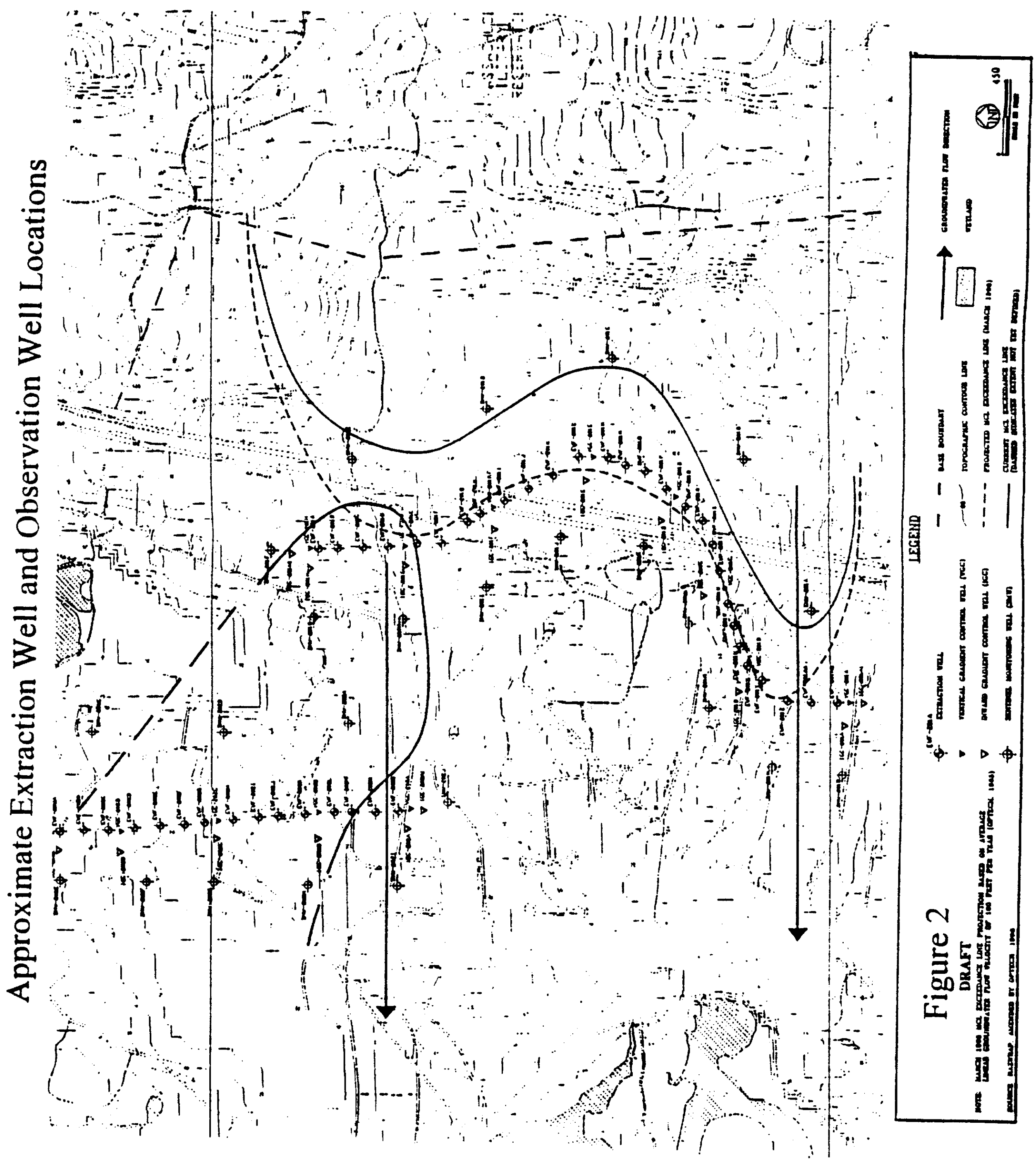

Figure 2 Locations of proposed extraction wells for the LF-1 plume
(from OpTech, 1996)

The fence line of wells at Route 28 is designed to capture the LF-1 contaminated groundwater that is flowing westward to Buzzards Bay. OpTech plume data which describe the spatial distribution of the contamination show that the leading edge of the plume has been detected passed Route 28. Since the proposed containment strategy will not capture this leading edge termed the "toe" of the plume, the detached plume of contaminated groundwater is expected to continue its migration and discharge untreated into Buzzards Bay. This containment strategy of extraction wells installed along Route 28 was proposed due to potential disturbance to the freshwater-saltwater interface along the coastline if the extraction wells are installed at the leading edge of the plume.

OpTech (1996) also performed a preliminary risk assessment of future potential effects to human health and ecological systems from contaminants within the "toe" after the installation of the recommended plume containment system. The maximum cancer risk for adult residents in the towns of Bourne and Falmouth is 4.7E-04 if the containment strategy is implemented. This cancer risk to adult residents from the detached contaminated plume is also above the USEPA acceptable standard. The overall maximum HI for non-cancer risk from exposure to the detached plume is 3.3. HI above the acceptable USEPA standard of 1.0 poses non-cancer health risks to exposed residents. The cancer and non-cancer risks posed by the detached plume are also above the USEPA standards. A comparison of the two preliminary risk assessment results indicates that the proposed containment system for the LF-1 plume will reduce the maximum cancer and non-cancer risks, but these risks are still significant and above the acceptable USEPA standards.

1.1 Objectives and Scope

The current thesis is a component of a larger project of the Master of Engineering program that investigated the environmental impacts of and possible remediation actions for the LF-1 plume. The study included characterization of the site, modeling of the groundwater plume, risk assessment, analysis of public involvement in the remediation process, strategy for source containment, and designing a bioremediation solution. The results of that study are included in Appendix.

This thesis addresses the issue of the risks posed to public health and the coastal ecosystem from the contamination of quahog clams in the coastal harbors of western Cape Cod due to the potential discharge of the LF-1 plume into these harbors. The shallow tidal flats of Red Brook and Megansett harbors, where the contaminants are predicted to discharge, support a rich population of local shellfish species. Soft shell clams, quahogs (hard clams), oysters, bay scallops, surf clams, mussels, and conch from these harbors are routinely harvested by local commercial and recreational fishermen. The contamination data for the LF-1 plume reported by OpTech (1996) show that the plume composed of heavy metals and volatile organic chemicals (VOC) is currently discharging into the Red Brook Harbor. In addition, the CDM Federal (1995) plume models predict that the contaminants of the LF-1 plume are likely to discharge into Red Brook and Megansett harbors in the near future.

Since heavy metals in the LF-1 plume have been detected above the background concentration levels for the MMR and shellfish have been shown to bioaccumulate metals in their body tissue, the discharge of the plume into the coastal harbors along the shoreline poses risks to the shellfish population of these areas as well as to consumers of shellfish. The objective of this thesis is to determine if any risk to public health and the coastal ecosystem exists from the potential bioaccumulation of heavy metals in quahog clams at Red Brook and Megansett harbors of Buzzards Bay. The current analysis will only consider contamination from metals that are detected in the LF-1 plume.

The assessment of risks posed to human health from consumption of tainted quahogs is performed using the standard USEPA risk assessment methodology. This methodology is detailed in the *Risk Assessment Handbook* prepared by Automated Sciences Group et al. (1994) for the MMR. The methodology and data section of chapter 3 will give a brief overview of this assessment methodology. In addition to posing undesired health effects on consumers of seafood, the potentially contaminated quahogs are also likely to threaten the populations of marine species that feed on them as a food source. The possible disruptive effects to the coastal ecosystem from the contamination of shellfish populations in the shoreline areas is only qualitatively analyzed due to a lack of data.

Chapter 2 Background and Site Description

2.1 Western Cape Geography and Land Use

The MMR is located in the northwestern portion of Cape Cod and covers an area of approximately 30 square miles (ABB, 1992). Figure 3 contains maps of the region and the reservation. Military use of the MMR began in the early 1900's and may be categorized as mechanized forces training and military aircraft operations. Since commencement of military operations, the base has seen use by several branches of the armed services, including the United States Air Force, Army, Navy, Coast Guard, and the Massachusetts Air National Guard. Operations by the Air National Guard and Coast Guard are ongoing.

The source area of the present plume study, the main base landfill, is about 10,000 feet from the western and southern MMR boundaries and occupies approximately 100 acres. The landfill has operated since the early 1940's as the primary waste disposal facility at MMR (CDM Federal, 1995). Unregulated disposal of waste at the landfill continued until 1984, at which time disposal began to be regulated by the Air National Guard.

Waste disposal operations at the landfill took place in a natural kettle hole and five distinct disposal cells as shown in Figure 4. These cells are termed the 1947, 1951, 1957, 1970, post-1970, and kettle hole. The date designations indicate the year in which disposal

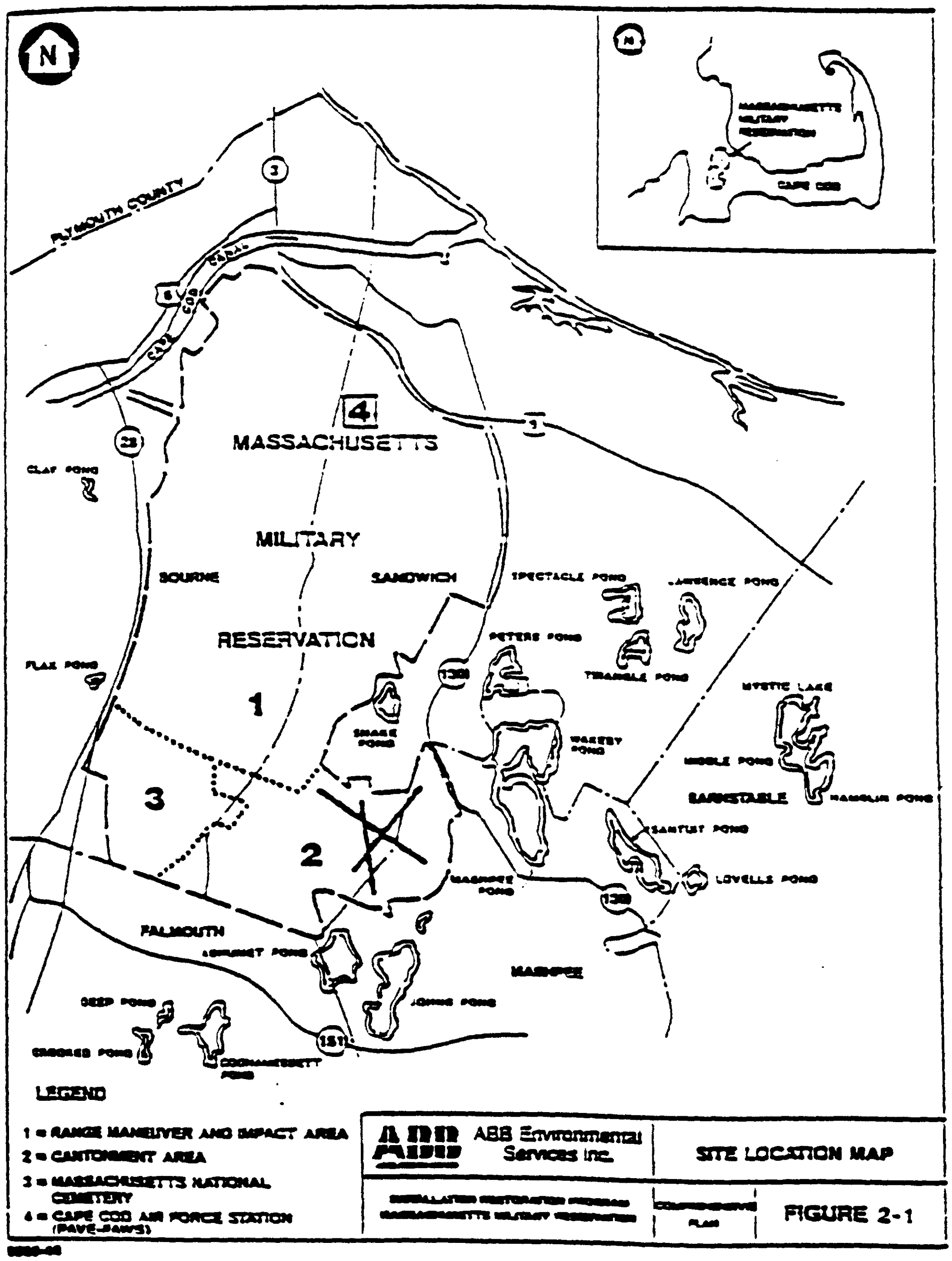

Figure 3 Site location map
(from ABB, 1992)

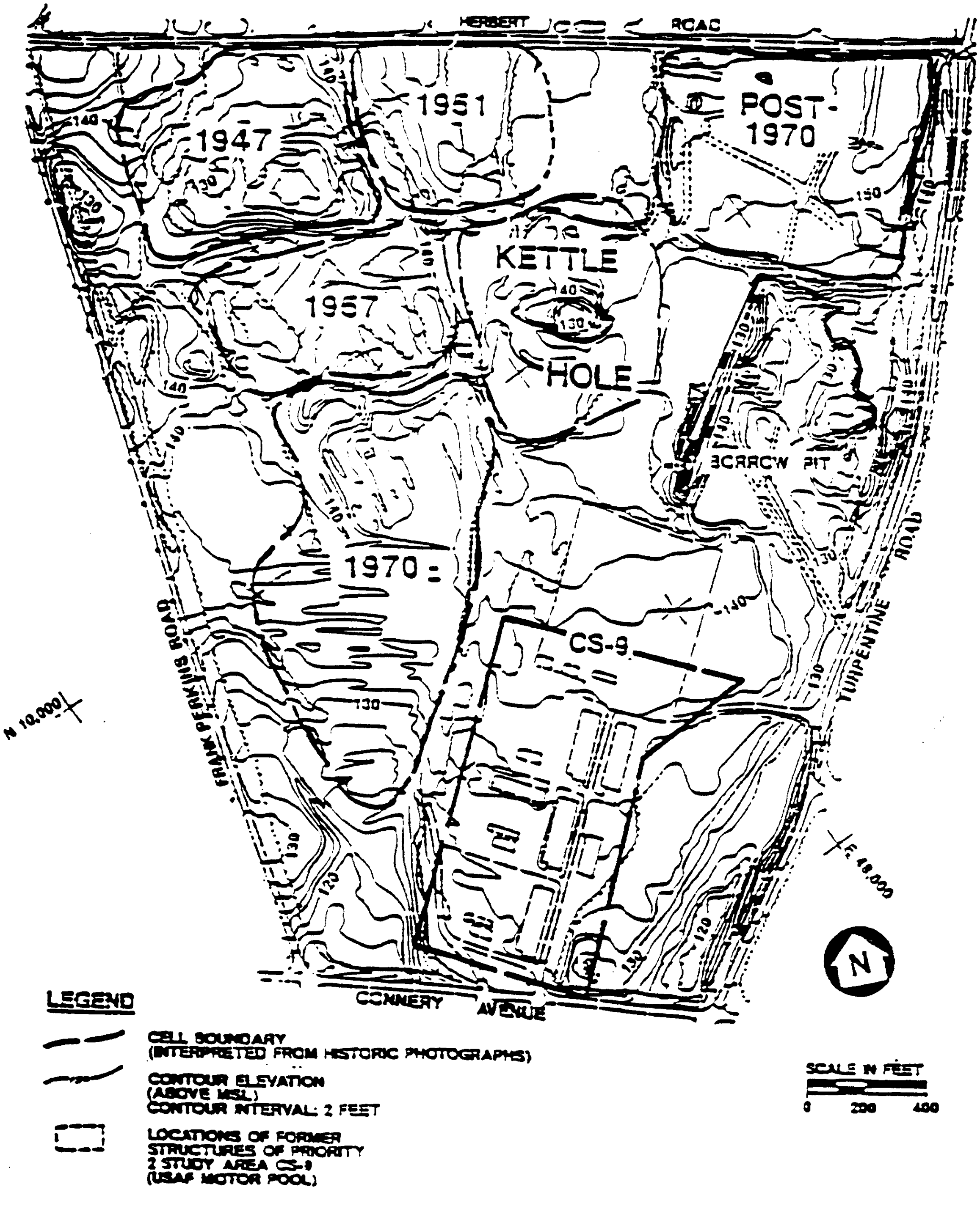

Figure 4 Photogrammetric map of landfill layout
(from ABB, 1992)

operations ceased at that particular cell. Accurate documentation of the wastes deposited in the landfill are not available. The wastes may include any or all of the following: general refuse, fuel tank sludge, herbicides, solvents, transformer oils, fire extinguisher fluids, blank small arms ammunition, paints, paint thinners, batteries, DDT powder, hospital wastes, municipal sewage sludge, coal ash, and possibly live ordinance (ABB, 1992). Wastes were deposited in linear trenches and covered with approximately 2 feet of native soil. Waste depth is uncertain, but estimated on average to be approximately 20 feet below the ground surface. In 1990, waste disposal at the landfill was ceased. A plume of dissolved chlorinated volatile organic compounds, primarily tetrachloroethylene (PCE) and trichloroethylene (TCE), has developed in the aquifer downgradient of the landfill.

The four towns of interest on the western Cape Cod are Bourne, Sandwich, Mashpee, and Falmouth. According to the 1994 census, the total population of this area is 67,400. The area is mostly residential with some small industry. A significant amount of economic activity is associated with restaurants, shops, and other tourist related industry. The total population of Cape Cod is estimated to triple in summer when tourists and summer residents make up the majority of the population. The total base population is estimated to have doubled in the last twenty years. Cape Cod has been one of the fastest growing areas in New England. In 1986, 27% of the economic activity was attributed to retirees; tourism accounted for 26%; seasonal residents, 22%; manufacturing, 10%; and business services (fishing, agriculture, and other), 15%; (Cape Cod Commission, 1996). The economy is currently experiencing a shift from seasonal to year-round jobs.

2.2 Climate

The Cape Cod climate is categorized as a humid, continental climate. Average wind speeds range from 9 mph from July through September to 12 mph from October through March. Precipitation is fairly evenly distributed with an average annual precipitation of approximately 47 inches. Surface runoff was observed to be minimal. Approximately 40% of the precipitation infiltrates through the unsaturated layer and enters the groundwater system (CDM Federal, 1995).

2.3 Geology

The Cape Cod Basin consists of material deposited as a result of glacial action during the Wisconsinian stage between 7000 and 80,000 years ago. Advancing glaciers from the north transported rock debris gouged from the underlying bedrock to the present most southern point at Martha's Vineyard and Nantucket Island. The glacial action also resulted in a thin layer of basal till being deposited over the bedrock. The entire sedimentation process occurred as a sequence of glacial deposition, erosion and re-deposition. In later periods, the glaciers melted, receded, and reached a stagnation point near the western and northern shores of Cape Cod. The remaining glacial till was deposited there and formed the Buzzards Bay and Sandwich moraines. The present day Sandwich Moraine is thought to be of glacio-tectonic origin, due to pro-glacial sediments being thrusted over older morainal deposits during a readvance of the Cape Cod Bay glacier (Oldale, 1984).

The regional geology in the LF-1 study area can be classified into three main sedimentary types. These are the Buzzards Bay and Sandwich moraines (BBM and SM), the Mashpee Pitted Plain (MPP) and the Buzzards Bay Outwash (BBO). The geographic distribution of these materials is depicted in Figure 5. The MPP consists of stratified coarse to fine grained sands that were transported from the melting Buzzards Bay and Cape Cod Bay ice sheets, and deposited over a bed of fine-grained glacio-lacustrine sediments and basal till. The general trends of the grain size in the glacial outwash deposits are coarsening upward and fining north to south. The thickness of the coarse material decreases north to south, as the distance from the outwash source increases.

The morainal sediments were deposited directly as the ice-sheets melted. Thus, these deposits are not stratified like the MPP glacial outwash and are thought to occur in layers of poorly sorted sediment-flow deposits and finer till material. These sandy sediments overlie a fining sequence of sand, silt, clay and basal till. The unsorted glacial till that comprise the BBM ranges in size from boulders to fine clays. This complex heterogeneity leads to wide variations in observed hydrogeological parameters in the

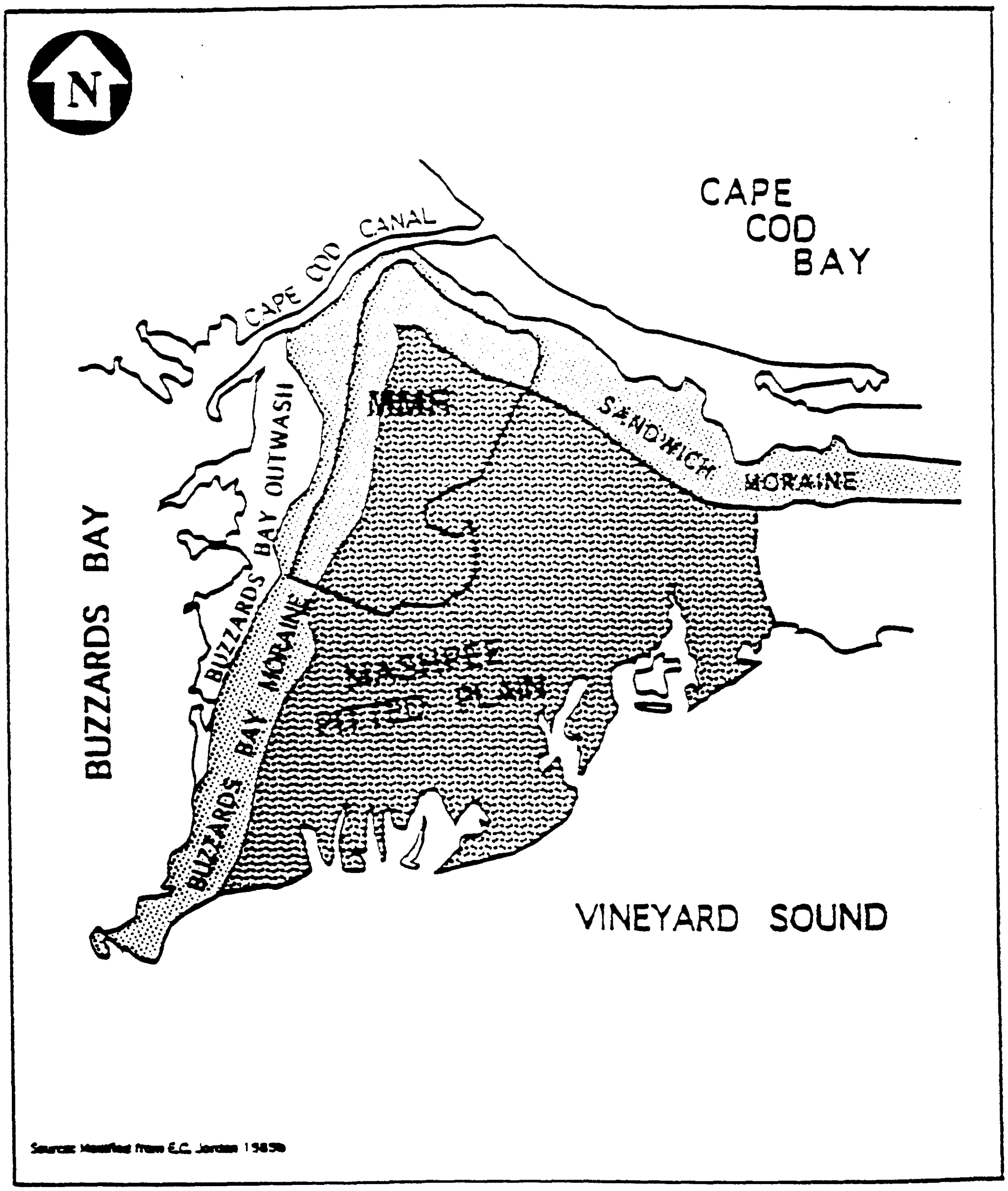

Figure 5 West Cape Cod glacial deposits
(from CDM Federal, 1995)

moraine. A general trend of fining in material size results in lower hydraulic conductivities (LeBlanc, 1986).

The Buzzards Bay Outwash (BBO) was deposited as a result of sedimentation between the retreating ice sheets and the newly deposited Buzzards Bay Moraine. BBO sediments are generally sand and gravel, and are considered to be stratified in the same manner as the MMP outwash, with a general trend of fining downwards.

The geologic structure described above lies atop a Paleozoic crystalline bedrock. The bedrock contours range in depth from 70 to 500 feet below sea level (Oldale, 1984). The bedrock is of a much lower hydraulic conductivity than the surrounding sediments, and thus acts as an impermeable barrier to groundwater flow and forms the bottom boundary of the Cape Cod aquifer.

2.4 Groundwater System

Cape Cod is underlain by a large, unconfined groundwater flow system. This phreatic aquifer has been designated a sole source aquifer by the USEPA. The aquifer is divided into six flow cells according to the hydraulic boundaries of the flow system. The MMR and LF-1 plume are located in the west Cape flow cell, the largest of the six flow cells. The aquifer system and water table contours in the west Cape region are depicted in Figure 6.

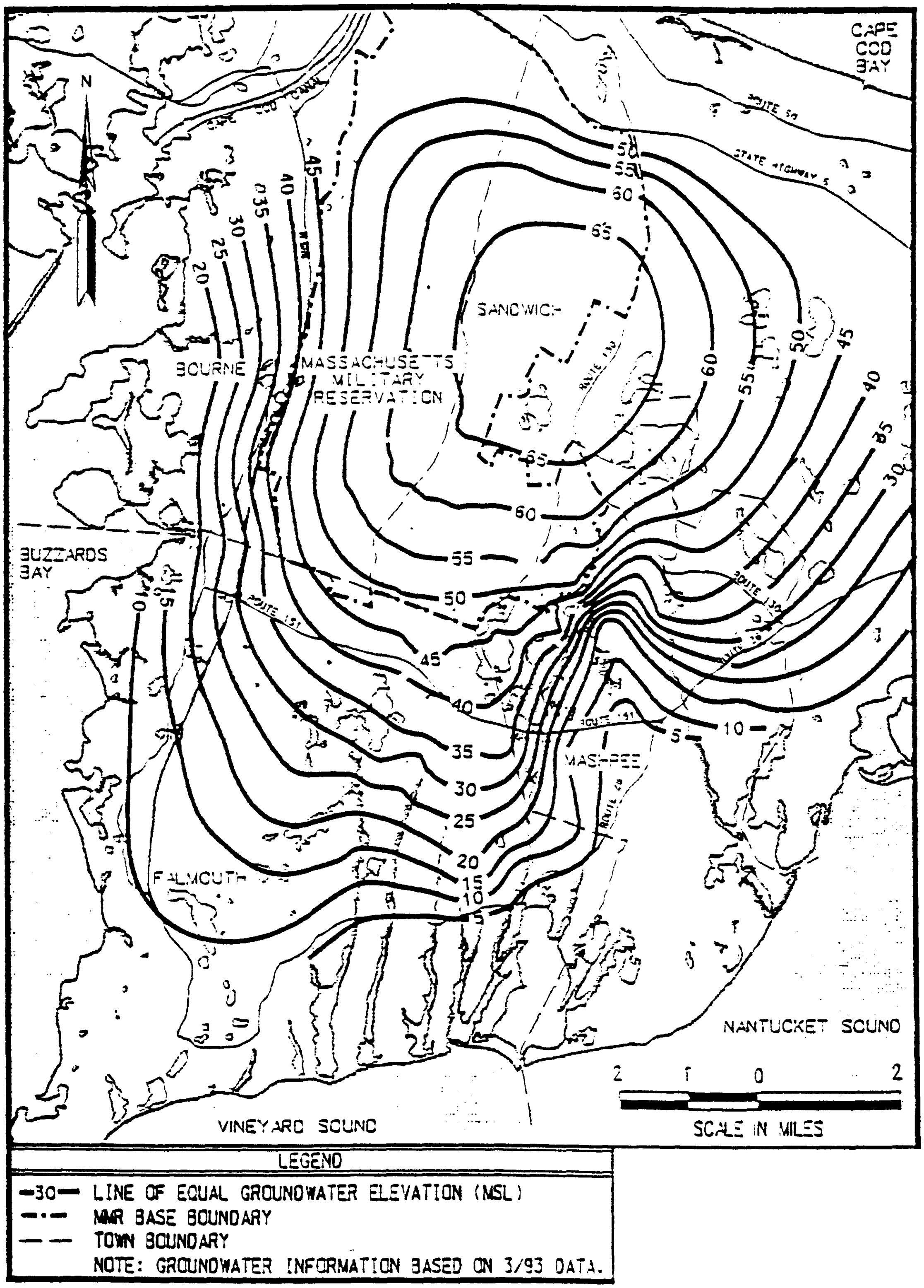

Figure 6 MMR groundwater contour map
(from Automated Sciences Group et. al., 1994)

The water table in this region occurs at a depth of 40-80 feet below the ground surface. Surface water is also present in the study area as intermittent streams in drainage swales and more importantly as ponds in kettle holes on the Mashpee Pitted Plain. However, there are a few large kettle ponds that can significantly influence the flow regime near the landfill site and the plume. Cranberry bogs can also occur at surface discharges of groundwater, but it is believed that the cranberry bogs west of the LF-1 site are underlain by localized perched water tables, and thus hydrologically disconnected from the larger aquifer system (CDM Federal, 1995). In addition, it is unlikely that contaminants from the LF-1 plume will discharge into these cranberry bogs since the plume is found deep within the aquifer below the bogs.

2.4.1 Vertical Hydraulic Gradients

Vertical gradients that have been calculated for the LF-1 site are very small. Most gradients calculated in the hydrologic investigations were below the survey accuracy threshhold. Significant upward vertical gradients do exist where groundwater discharges into large ponds and near coastal areas where the aquifer discharges into the ocean. Small downward gradients of about 10^{-3} to 10^{-4} ft/ft are observed throughout the rest of the study area (CDM Federal, 1995). Such vertical gradients generally indicate upward flow near the shoreline and surface water bodies and downward flow elsewhere.

2.4.2 Horizontal Hydraulic Gradients

Groundwater flow in the region is driven mostly by horizontal gradients. These can be measured by dividing a groundwater elevation contour interval by the horizontal distance between the contours. The latter value can be estimated from a contour map similar to Figure 6. Horizontal gradients calculated for the LF-1 study area using February 1994 water levels range from 1.3×10^{-3} to 6.8×10^{-3} ft/ft (CDM Federal, 1995). These gradients are observed to steepen from the LF-1 source area westwards.

2.4.3 Seepage Velocity

Calculated seepage velocities in the LF-1 study area indicate that advective contaminant transport takes place at velocities ranging from 0.10 ft/day to over 3 ft/day. Since seepage velocity is a function of hydraulic conductivity, the differential permeabilities of the various sediment types strongly influence the calculation of seepage velocities at this site. An estimate of the seepage velocity of contaminants using the observed LF-1 plume migration distance and time yielded an average seepage velocity of 0.9 ft/day (CDM Federal, 1995).

2.5 MMR's Listing on the National Priorities List

The MMR is one of 1,236 sites that have been placed on the NPL by the USEPA. NPL sites are those to which the EPA has given particularly high human health and environmental risk ranking. Rankings are determined from an evaluation of the relative risk to public health and the environment from hazardous substances identified in the air, water and natural surroundings of the site. Once placed on the NPL, Superfund monies are used to finance remedial cleanup of the sites. Additional funding may also come from potentially responsible parties, those individuals and organizations whose activities have resulted in contamination.

2.6 Present Activity

Due to the health and environmental risks which have been attributed to activities at the MMR, federal activity is underway to further quantify and reduce, to the extent required, the risk imposed upon human health and the environment. As part of remediation operations at the MMR, several of the landfill cells have recently been secured with a final cover system. These cells include the 1970, the post-1970, and the kettle hole. The remaining cells (1947, 1951, and 1957) have collectively been termed the Northwest Operable Unit. Remedial investigations as to the necessity of a final closure system for these cells is ongoing. Other ongoing activities associated with the LF-1 site include further plume delineation, groundwater modeling, and design of a plume containment and possible remediation system.

Chapter 3 Human Health and Ecological Risk Assessment

3.1 Introduction

Risk assessments are performed for hazardous waste sites in order to measure their potential for harm to humans and the natural ecosystem. It is a complex process filled with enormous scientific uncertainties. Scientists and risk assessors still question the toxicological data of the majority of the suspected chemicals that can potentially have harmful effects on humans and other living organisms. In addition to the uncertainties surrounding the toxicity of the chemicals, exposure assessments in risk calculations also include a high degree of estimation.

Risk is a function of both the probability of the adverse event occurring and the magnitude of the event. Thus, risk events with equal products of probability times magnitude are regarded as equivalent. People are generally likely to accept risks of low damage and high probability than risks of high damage and low probability. An example of this risk aversity is the wide acceptability of risk in automobile travel compared with the intolerance to risk in airplane travel.

Risks posed to humans from man-made hazardous wastes are incremental risks above the natural background level. The incremental risk is the excess risk that is imposed on individuals by an external agent. Background hazard is the risk that the individual is exposed to from the natural environment and from its own idiosyncrasies in the absence of incremental risk. Thus, the total risk for each individual is the sum of the background and incremental risks. Table 1 shows the background risk for certain actions and behaviors as well as the maximum incremental risk allowed by the USEPA risk standards.

Action	***Risk***
Background	
Cigarette Smoking, one pack per day	0.25
All Cancers	0.22
Death in a motor vehicle	0.02
Homicide	0.01
Radon in Homes, cancer deaths	0.003
Sea level background radiation, cancers	0.001
Incremental	
USEPA maximum contaminant level risk	0.0000001 - 0.0001

(adapted from Masters, 1991)

Table 1 The background and incremental risks in the United States

The USEPA has developed a standard methodology to assess risks posed to humans and the natural environment from hazardous waste sites. This systematic technique was developed to ensure that risk assessed from different waste sites can be compared uniformly. A preliminary risk assessment for potentially exposed adult residents and workers and the natural ecosystem has been completed for the LF-1 plume at the MMR (OpTech, 1996, CDM Federal, 1995). These preliminary studies did not assess the potential risk to human health from consumption of contaminated shellfish nor the risk posed to the coastal ecosystem from the potential decline of the shellfish population due to the likely discharge of the LF-1 plume into Red Brook and Megansett harbors of Buzzards Bay..

Brown et al. (1988) reported that seafood containing trace amounts of one or more of the following metals: arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc, and their compounds can potentially be hazardous to human health. Since all these metals and a few others have been detected within the LF-1 plume, the discharge of the plume contaminants into areas where shellfish are harvested for human consumption poses risks to the population that consistently obtain seafood from these areas. This chapter will characterize the risk posed to consumers of shellfish contaminated with metals by utilizing the standard USEPA risk assessment methodology.

3.2 Methodology and Data

USEPA's standard risk assessment methodology consists of four major steps:

1. Hazard Identification
2. Exposure Assessment
3. Toxicity Assessment
4. Risk Characterization

3.2.1 Hazard Identification

In quantifying the risk from a contaminated site, a list of hazards present at the site is initially determined. From the identified hazards, a list of chemicals of concern (COC) (i.e., chemicals that are likely to pose potential risk to human health and ecological systems) is assembled. After COCs are defined, a criteria to narrow down the list of COC to a few chemicals that represent the majority of the risk from the site is composed. The risk posed by this short list of COCs is typically 99% of the total incremental risk from the contaminated site. The criteria to reduce the COC list commonly involve the selection of chemicals that are most toxic to humans and other species, most persistent and mobile in the environment, and most widely distributed. Other criteria include chemicals that contribute to the highest exposure and that have the highest concentrations.

The hazards of the LF-1 plume are the metals and volatile organic compounds (VOC) that have been detected in the groundwater aquifer downgradient from the landfill. Since shellfish have been determined not to bioaccumulate VOCs (OpTech, 1996), only inorganic metals are considered as chemicals of concern. The COCs for the current risk analysis are aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, copper, cyanide, iron, lead, manganese, mercury, nickel, vanadium, and zinc. The metals that pose the most significant cancer risk to humans through consumption of tainted quahogs are arsenic and beryllium. A majority of non-cancer risk is derived from iron, aluminum, arsenic, chromium, and manganese.

3.2.2 Exposure Assessment

Exposure assessment consists of determining the population that is likely to be exposed to the contaminants, the concentration at the exposure point, exposure pathways, and exposure duration and intake.

The 1990 and projected populations for the two Cape Cod towns, Bourne and Falmouth, that are affected by the LF-1 plume are presented in Table 2.

	1990 Federal Census	1990 Summer Population	Projected 2020 Population	Projected 2020 Summer Pop.
Bourne*	16,064	24,214	19,642	27,792
Falmouth	27,960	69,900	33,867	82,297

Source: Office of Water Resources, Mass. DEM, 1994
* Population of MMR is included in Bourne figures.

Table 2 Population data for the towns of Bourne and Falmouth

The maximum metal exposure concentration for quahogs is assumed to be the maximum concentration detected in the LF-1 plume. This is a conservative estimate since there will most likely be dilution of the concentration when the metals discharge into the bay. Nevertheless, this assumption is partially valid since the clams reside in the sediments and could possibly uptake pore water of the sediments. The detected maximum concentrations of the metals in the LF-1 plume are presented in Table 3.

	Max. Conc.[a] (ug/l)	Max. Conc.[b] (ug/l)
Aluminum	20900	10,200
Antimony	NA	2.6
Arsenic	3.5	8.4
Barium	400	107
Beryllium	3.6	1.1
Cadmium	2	2
Chromium#	54.2	66.3
Copper	48.7	28.2
Cyanide	16.4	NA
Iron	134,000	24,000
Lead	27.8	9.8
Manganese	5040	824
Mercury	0.3*	0.3*
Nickel	24.4	184
Vanadium	33	41
Zinc	262	184

Source: a CDM Federal (1995)
b OpTech (1996)

Chromium (VI) values are used

* Maximum dissolved concentration

NA Not available

Table 3 Maximum detected concentrations

The major exposure pathway considered for humans is ingestion. Ingestion is defined as the oral consumption of contaminants. The primary intake pathway of contaminants for quahogs is also considered to be ingestion.

The exposure duration for quahogs is estimated at 2.5 years, the average life of a quahog (Hickey, 1996). With this exposure time, metal concentration in quahogs can be estimated by

$$C_{quahog} = \frac{C_{plume} \times AL \times PR \times 365 \text{ days/yr} \times 0.10}{10^3 \text{ mg/ug} \times W_{quahog}}.$$

C_{quahog} = Maximum metal concentration in quahogs (mg/kg of wet weight)
C_{plume} = Maximum metal concentration in the plume (ug/l)
AL = Average life of quahogs = 2.5 years
PR = Pumping rate of quahogs = 1.75 l/day
W_{quahog} = Average wet tissue weight of a quahog = 0.030 kg

The uptake rate of quahog clams is assumed to be 10 percent. This figure is estimated from a review of the literature (Bordin et al., 1992; Paez-Osuna et al., 1993; Peerzada et al., 1992; Sadiq et al., 1992). The pumping rate and average wet tissue weight of quahogs are estimates by Hickey (1996). The maximum concentration of each metal in quahog clams is presented in Table 4.

The human exposure duration is assumed to be 70 years. Intake of metals by humans through consumption of contaminated shellfish is approximated by

$$\text{Intake (mg/kg-day)} = \frac{C_{quahog} \times \text{Consumption}}{W_{human}}.$$

Intake = Average daily intake of metals
Consumption = Average human daily consumption of quahogs = 0.002 kg/day
W_{human} = Average adult body weight = 70 kg

The average daily human consumption of quahogs is estimated from Brown et al. (1988). This daily average is derived from an annual consumption of seafood by an average consumer. The intake of quahog clams is assumed to be 10 percent of total annual seafood consumption. The average adult body weight is derived from the *Risk Assessment Handbook.* The potential human intake of metals from contaminated quahogs is shown in Table 4.

	$^{a}C_{quahog}$ (mg/kg)	$^{b}C_{quahog}$ (mg/kg)	Human Intake[a] (mg/kg-day)	Human Intake[b] (mg/kg-day)
Aluminum	111376	54356	3.18217	1.55302
Antimony	0	14	0	0.0004
Arsenic	19	45	0.00053	0.00128
Barium	2132	570	0.0609	0.01629
Beryllium	19	6	0.00055	0.00017
Cadmium	11	11	0.0003	0.0003
Chromium	289	353	0.00825	0.01009
Copper	260	150	0.00741	0.00429
Cyanide	87	0	0.0025	0
Iron	714086	127896	20.4025	3.65417
Lead	148	52	0.00423	0.00149
Manganese	26858	4391	0.76738	0.12546
Mercury	2	2	4.6E-05	4.6E-05
Nickel	130	252	0.00372	0.0072
Vanadium	176	218	0.00502	0.00624
Zinc	1396	981	0.03989	0.02802

a using CDM Federal (1995) plume concentrations
b using OpTech (1996) plume concentrations

Table 4 Maximum metal concentrations in quahogs and human intake of metals

Since the maximum concentrations of aluminum and iron detected in the LF-1 plume were much higher than the background levels, the potential concentrations of these metals in quahogs are also particularly high. It is unlikely that the quahog clams will accumulate these two metals to the concentration levels calculated in Table 4. The concentration levels in quahogs are likely to reach an upper limit for these two metals before arriving at the concentration figures shown in Table 4. The calculated concentration levels are worst-case assumptions. The figures assume that a quahog clam will retain 10 percent of the maximum metal concentration that it ingests within its lifetime of 2.5 years. Thus, the potential intake of aluminum and iron by humans is a conservative estimate.

3.2.3 Toxicity Assessment

The toxicity assessment consists of the determination of deleterious health effects to humans from chemical compounds. USEPA has separated chemicals and metals into carcinogenic and non-carcinogenic compounds for humans. To determine the toxicity of chemicals to humans, epidemiological and animal test studies are conducted. From the results of these studies, dose-response curves for each chemical or metal is constructed. Dose-response relationships plot the percentage of a test population exhibiting a predetermined response over a dose range. The median lethal dose is the dosage where 50% of the population shows a predetermined response. Figure 7 shows a theoretical dose-response relationship.

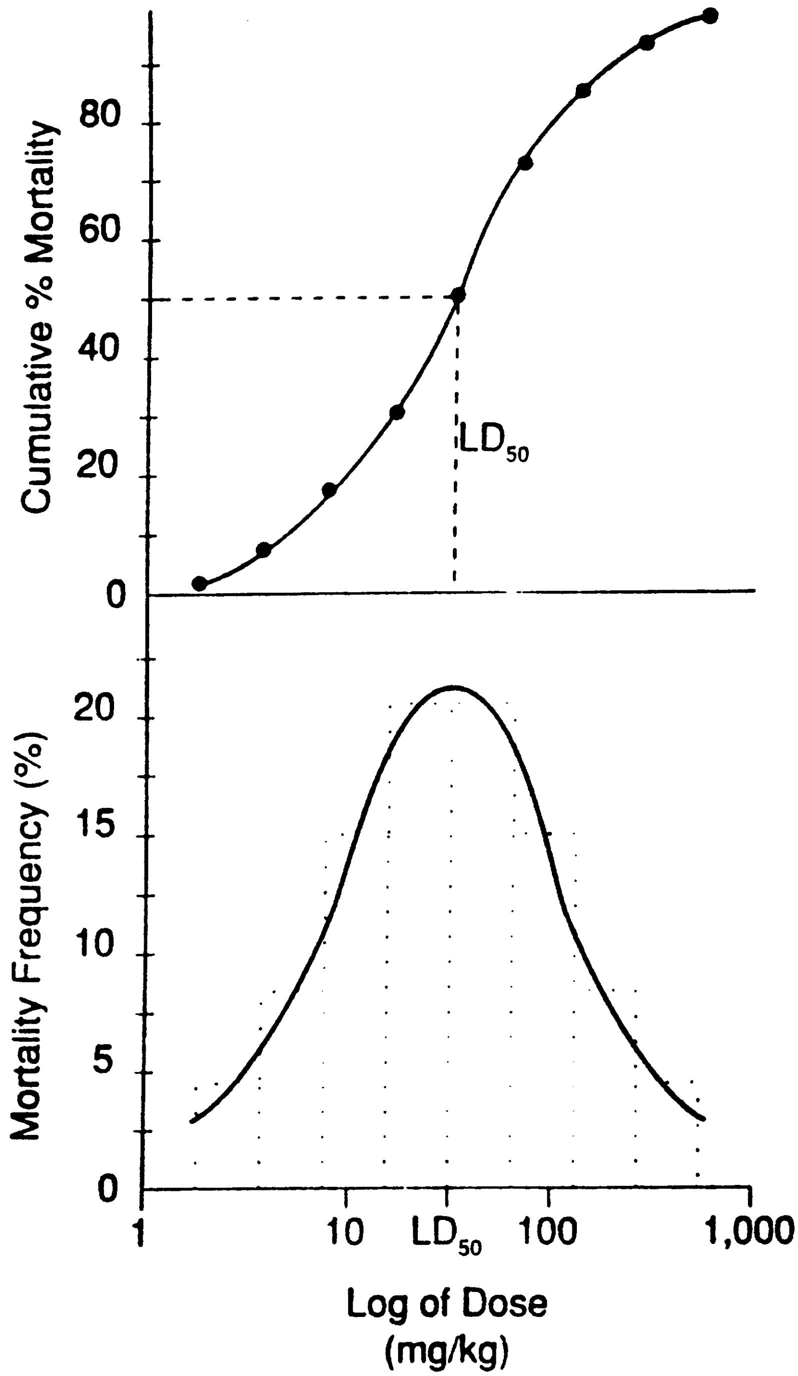

Figure 7 Theoretical dose-response relationship
(from LaGrega, 1994)

The toxicity data for each metal are obtained from the *Risk Assessment Handbook*. The toxicity values obtained are the reference dose values (RfDs) for non-carcinogenic effects and cancer slope factors (CSFs) for carcinogenic effects. The oral CSFs and RfDs as well as the EPA cancer classifications for all COCs are presented in Table 5.

	Oral CSF	Oral RfD	USEPA Cancer Class[a]
Aluminum	NA	1	NA
Antimony	NA	0.0004	ND
Arsenic	1.75	0.0003	A
Barium	NA	0.07	ND
Beryllium	4.3	0.005	B2
Cadmium	NA	0.001	B1
Chromium	NA	0.005	A
Copper	NA	NA	D
Cyanide	NA	0.02	D
Iron	NA	0.5	NA
Lead	NA	NA	B2
Manganese	NA	0.14	D
Mercury	NA	0.0003	D
Nickel	NA	0.02	ND
Vanadium	NA	0.007	ND
Zinc	NA	0.3	D

Source: *Risk Assessment Handbook*
Notes: (a) See Table 6
NA = Not Available

Table 5 Oral cancer slope factors, oral reference doses, and USEPA classification of cancer class for metals

The cancer slope factor is the slope of the dose-response curve for a carcinogen. It is the 95 percent upper confidence limit of the probability of carcinogenic effect (LaGrega, 1994). These slope factors are used to determine the carcinogenic risk to humans from a chemical and are expressed as a probability of developing cancer from lifetime exposure. Excess lifetime cancer risk is calculated by multiplying the lifetime average daily intake with the cancer slope factor. For exposure to multiple carcinogens, the risk is the sum of the individual risks.

The USEPA has developed a ranking order for carcinogens using the most up-to-date data. The categories for the carcinogens are presented in Table 6.

A	=	human carcinogen
B1	=	probable human carcinogen
B2	=	probable human carcinogen
C	=	possible human carcinogen
D	=	not classifiable as to human carcinogenicity
PD	=	pending
ND	=	no data

Table 6 USEPA classification system for carcinogens
(adapted from Harte et al., 1991)

A chemical is given a Class A identification only if there is sufficient epidemiological data to support a cause-effect relationship. B1 classification is based on a limited human data and sufficient evidence from animal studies, while B2 classification indicates that there is sufficient data from animal studies but inadequate epidemiological data. If the chemical has only a limited animal test evidence with no epidemiological data, then it is given a Class C ranking. Chemicals are classified as Class D when there is inadequate data from animal or epidemiological studies.

To assess human health effects from non-carcinogenic chemicals, the USEPA's established reference doses for non-carcinogens are applied. The reference dose determines the allowable estimated daily intake for a chemical that will not cause adverse health effects. The ratio of the estimated daily intake of a contaminant to its reference dose is defined as the hazard quotient of a chemical. In risk analyses for non-carcinogenic effects, hazard quotients of COCs are summed to arrive at a hazard index (HI) for the site. A hazard index of greater than one can potentially cause damaging non-cancer health effects. The HI approach developed by USEPA assumes that particular organs of the body are exposed simultaneously to multiple chemicals and that the effects from exposure are additive.

3.2.4 Risk Characterization

Risk characterization is the final step in the risk assessment process. Human intake of chemical of concerns for each pathway of exposure are combined with the toxicity values of the chemicals to determine the risk. Cancer and non-cancer risks are calculated separately for the exposed population since toxicological mechanisms are different for each effect. In conservative risk calculations, maximum contaminant concentrations are used as exposure concentrations. The results of the current risk characterization are presented in the results section of this chapter.

3.3 Characterization of Baseline Ecology

Buzzards Bay is characterized as a well-mixed tidal estuarine system. The average depth in the central basin is 50 feet while the approximate width and length are 8 and 28 miles, respectively (CDM Federal, 1995). Marine aquatic organisms inhabiting the bay include flounders, bass, eels, scallops, lobsters, crabs, mussels, quahogs, and oysters. In addition, avian species of osprey, gulls, and terns also reside in the area.

Federal endangered or threatened species that may possibly inhabit the Buzzards Bay area include the blue whale, bowhead whale, finback whale, gray whale, humpback whale, right whale, sei whale, sperm whale, and the short-nose sturgeon (CDM Federal, 1995). Furthermore, the loggerhead sea turtle, a threatened species in Massachusetts, and the Atlantic Ridley and leatherback sea turtles, state endangered species, inhabit the Buzzards Bay waters during the summer period. The piping plover, a federally endangered species, also frequently nests and feeds on the beaches of Buzzards Bay.

3.4 Results

3.4.1 Human Health Risk from Ingestion of Contaminated Shellfish

The results of maximum cancer and non-cancer risk assessment from consuming tainted quahogs over a lifetime for each metal contaminant are calculated in Table 7. Using the CDM Federal (1995) data, the total maximum cancer risk from consumption of contaminated quahogs is 3.3E-03. When maximum detected metal concentrations from OpTech (1996) is used in the assessment, a total maximum cancer risk of 3.0E-03 is found. The cancer risk for humans from consumption of tainted quahogs is derived from only two metals - arsenic and beryllium - since these are the only metals with published cancer slope factors. The USEPA's acceptable cancer risk standard is 1.0E-06.

	Cancer Risk[1]	Cancer Risk[2]	Hazard Index[1]	Hazard Index[2]
Aluminum	NA	NA	3.18217	1.55302
Antimony	NA	NA	0	0.98967
Arsenic	0.00093	0.00224	1.77633	4.2632
Barium	NA	NA	0.87004	0.23274
Beryllium	0.00236	0.00072	0.10963	0.0335
Cadmium	NA	NA	0.30451	0.30451
Chromium	NA	NA	1.65047	2.01893
Copper	NA	NA	NA	NA
Cyanide	NA	NA	0.12485	0
Iron	NA	NA	40.8049	7.30834
Lead	NA	NA	NA	NA
Manganese	NA	NA	5.48126	0.89614
Mercury	NA	NA	0.15226	0.15226
Nickel	NA	NA	0.18575	1.40077
Vanadium	NA	NA	0.71778	0.89179
Zinc	NA	NA	0.13297	0.09338

Notes: 1 Derived by using CDM Federal (1995) plume data
2 Derived by using OpTech (1996) plume data
NA = Not available

Table 7 Maximum cancer and non-cancer risk from each metal

The overall maximum hazard index (HI) for non-cancer risk from potential exposure to contaminated quahogs is 55.5 and 19.1, when CDM Federal (1995) and OpTech (1996) data, respectively, are used in the assessment. The USEPA's acceptable HI standard for non-cancer risk is 1.0. Calculated HI that is above the USEPA standard poses possible non-cancer deleterious health effects to exposed populations. The total maximum cancer and non-cancer risks from contaminated quahogs are summarized in Table 8. Both the cancer and non-cancer risks are above the USEPA standards.

	Maximum Cancer Risk	Maximum Hazard Index
CDM Federal Data	3.3E-03	55.5
OpTech Data	3.0E-03	19.1

Table 8 Total maximum cancer and non-cancer risks from consumption of tainted quahogs

Assumptions

A major assumption of the current risk estimation is that the maximum metal concentrations detected in the plume will discharge into the coastal harbors without any retardation. This scenario is unlikely since the metals will likely sorb on to the fine sediments of the aquifer, and thus retarding the flow of metals towards Buzzards Bay. A dilution factor of one (no dilution) was also assumed in the calculation for the period after the discharge of the plume into the harbors. This factor was used in the computation of the maximum potential risk. In real physical settings, dilution and dispersion are likely to play major roles.

Uncertainties

The single largest source of potential error in the current risk assessment is the uncertainty that surrounds the toxicological data of metals for humans. This uncertainty stems from the lack of data as well as difficulties in conducting epidemiological and animal test studies. Epidemiological studies are extremely insensitive in detecting health effects from low levels of exposure. Experimental animal studies are also commonly conducted at high dose levels to ensure that results are statistically significant. In addition to the difficulties in obtaining reliable data, the debate between the traditional toxicologists and the emerging molecular biologists over the existence of threshold levels for chemicals in biological species adds yet another uncertainty to the established toxicological data. The traditional toxicologists believe that the duration and concentration of exposure must be large enough to cause some adverse irreversible response. On the other hand, the molecular biologists claim that even minute exposure to toxins can cause deleterious effects to humans.

Due to the large uncertainty surrounding the toxicity data for humans, the USEPA utilizes safety factors in the risk calculations. The 95 percent confidence level built into slope factors for carcinogens ensures that the slope factor will be conservative 19 out of 20 times. For non-carcinogens, the USEPA's established safety factors are applied to the reference doses. Table 9 illustrates the areas of uncertainty and safety factors that are applied for non-carcinogens.

Area of uncertainty	*Safety factor*
Variation within a population	10
Extrapolation from animals to humans	10
Extrapolation from sub-chronic to chronic	10
Extrapolation from LOAEL to NOAEL	10
Modifying factor	1-10

Notes: LOAEL = Lowest Observed Adverse Effect Level
NOAEL = No observed Adverse Effect Level
(adapted from LaGrega, 1994)

Table 9 Safety factors applied to reference doses of non-carcinogens

In addition to the uncertainties surrounding the dose-response figures in risk calculations, numerous other factors also contribute to the complexity and uncertainty of risk estimation. One such factor is the systematic differences in the responsiveness of particular subgroups (e.g. children and elderly) within the exposed population to certain types of chemicals. Cofounding factors from multiple chemicals are also extremely difficult to determine. Occasionally, metabolites from physiological degradation of parent chemicals are more toxic to humans than the original chemicals. Due to the large uncertainties related to risk assessment, the results of the current analysis need to be interpreted critically and used with prudence.

3.4.2 Qualitative Assessment of Potential Ecological Effect

Since quahog clams are predicted to bioaccumulate metals, the discharge of the LF-1 groundwater plume into Red Brook and Megansett harbors can potentially have detrimental effects to the coastal ecological system. Quahogs are a primary food source for certain marine species that reside in the coastal harbors of Buzzards Bay. The contamination of the quahog clams will reduce its population thus triggering a decline in the populations of marine species that depend on quahogs as their sole food source. The decline in population of key species in the ecosystem can lead to an overall decline of the entire ecosystem.

The bioaccumulation of metals by quahog clams can also have detrimental effects on the ecosystem through another mechanism. Since quahog clams do not reside at the top of the coastal ecosystem food web, they are consumed by higher order species. In this process of nutrient transfer up the food chain, contaminants accumulated within lower food chain organisms are also transferred up the food web. Thus, tainted quahogs can potentially transfer accumulated metals to higher order food chain species. The bioaccumulation of metals in higher order organisms will lead to a decline in the population of the species resulting in the overall decline of the ecosystem as a whole.

Chapter 4 Conclusions

The current assessment of risk from consumption of potentially tainted quahogs from Red Brook and Megansett harbors shows that both cancer and non-cancer risks are significant, and above the USEPA standards. The USEPA risk standards are set at levels that adequately protect human health and the natural environment. The computed risks indicate that contaminated quahogs from the coastal harbors where LF-1 plume is predicted to discharge pose significant risks to consumers that consistently consume shellfish from these harbors. These current risk estimations are based on worst case assumptions. Thus, the risk is a conservative estimate and indicates the maximum potential risk to human health.

Even though the risks calculated are significant and above the USEPA standards, the accuracy of the assessment results is difficult to determine. The difficulty in quantifying the uncertainty of the results stems from the uncertainties that are imbedded in the toxicological data and the conservative assumptions that are made in the exposure assessment. When the uncertainties of the toxicological data and the exposure assessment are combined in the risk analysis, the potential uncertainty that will arise for the assessment result can not be easily quantified or interpreted. Thus, the degree of uncertainty that surrounds the current risk assessment results is unknown.

From these results, it is recommended that a monitoring program for shellfish harvested from Red Brook and Megansett harbors be implemented. In addition, more accurate exposure assessments such as comprehensive modeling of the plume and better estimation of dilution and dispersion factors in the harbors will improve the accuracy of the risk calculations. A laboratory analysis to determine the sorption of metals to aquifer sediments will also increase the precision of future assessments of risk. Furthermore, reductions in the uncertainty of the toxicological data will also improve the accuracy of future risk assessments.

Accurate estimations of risks are needed to better manage potential risks posed to human health and natural ecosystems. Large uncertainties in risk calculations can lead to perception of risk by the public that can either be well founded or incorrect. Risk assessments are only one of many factors including political, social, and economic, that need to be considered in the management of risk. Thus, effective communication of risk assessment results to the affected communities is crucial.

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Appendix

An Investigation of Environmental Impacts of the Main Base Landfill Groundwater Plume, Massachusetts Military Reservation, Cape Cod, Massachusetts

by

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May 1996

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1. Executive Summary

The Massachusetts Military Reservation (MMR), located at Cape Cod, Massachusetts, is currently listed on the National Priorities List of Superfund sites by the United States Environmental Protection Agency. This research project was undertaken to characterize the impacts of the groundwater contamination resulting from the Main Base Landfill (LF-1) and suggest possible remedial actions. Efforts were taken to characterize the state of the groundwater plume leaching from the landfill and the risks associated with the contamination. Groundwater modelling was utilized to predict the future movement of the plume. Current management approaches, as they relate to public perception and the involvement of the public in environmental restoration activities at the MMR, were also studied. In addition, possible remedial approaches, including source containment and bioremediation, were evaluated.

The large area of contamination emanating from the Main Base Landfill consists mainly of halogenated volatile organic compounds. The contaminant plume is heading west towards Buzzards Bay. The potential risks from the plume contamination were examined and found to be of significant concern, although highly uncertain. The uncertainty involved in risk assessment is one of the conditions that causes the public to perceive the risk as high. The analysis of public interaction in the environmental restoration process revealed concerns about the level of trust the public currently maintains in agencies which are responsible for regulating cleanup at the MMR. For the purpose of minimizing further contamination of the aquifer, a landfill final cover design for a portion of the uncapped area of the Main Base Landfill is proposed. Further, a bioremediation scheme for in situ removal of the plume contaminants is presented.

In summary:

* Site characterization showed
    - Drinking water resources exceeds EPA standards
    - Hydraulic conductivity can be characterized for modeling purposes by putting point values through a gaussian filter
* Modeling demonstrates that the moraine is the critical factor affecting plume migration
* Risk assessment modeling illustrates
    - An increase in relative cancer risk of 1.14E-04
    - Uncertainty in risk calculations is predominantly, if not almost entirely, dominated by the uncertainty of the EPA's cancer slope factors
    - potential risk from consumption of tainted shellfish is significant
* Remediation options can mitigate the impacts by
    - Capping the remaining landfill cells
    - Utilizing bioremediation as a potential cleanup tool
* Future approaches to the management of public interaction should
    - Involve the public early in the process
    - Strike an appropriate balance among MMR stakeholders.

2. Introduction

This report summarizes findings of a research project undertaken to characterize environmental impacts of groundwater contamination emanating from the Main Base Landfill at the Massachusetts Military Reservation (MMR), located at Cape Cod, Massachusetts. The United States Environmental Protection Agency placed the MMR on the Superfund National Priorities List (NPL) in 1989. This project report was submitted in partial fulfillment of the course requirements for the Master of Engineering Program in the Department of Civil and Environmental Engineering at the Massachusetts Institute of Technology (MIT).

Each individual on the project team researched a specific topic associated with the site. Individual findings were compiled as individual thesis reports in partial fulfillment of the Master of Engineering Degree requirements. A list of the thesis report titles can be found in the Appendix A of this document.

An extensive amount of data on contamination at the MMR has been collected and is maintained by the MMR Installation Restoration Program (IRP) office. The IRP acts as principal agent for the U.S. government on behalf of the MMR. Numerous reports have been generated for the IRP which include data observations and professional opinions. These reports are available for public review and are the principal source of information used for analysis in this report. A general assumption of this project research and subsequent report is that the analytical data which has been collected and reported in these documents are accurate.

This group project report both examines and offers opinions on the potential impacts of the MMR LF-1 on public health and welfare and how these effects can be mitigated. The scope of the research project includes site characterization and groundwater modeling, risk assessment, management of public interaction, study of source containment, and bioremediation technology. The underlying objectives of the report are as follows:

- Characterization of the site through evaluation of subsurface hydraulic conductivity
- Characterization of the landfill plume constituents, dimensions, and movement through use of existing data and groundwater modelling
- Evaluation of the potential cancer risk which materials identified in the groundwater present to people located near the landfill plume, as well as risks associated with ingestion of potentially contaminated shellfish
- Characterization of the management of public interaction surrounding base cleanup activities
- Protection of the Cape Cod groundwater aquifer from further contamination by source containment through the design of a landfill final cover system
- Design of a bioremediation scheme to remediate contaminated groundwater

3. Characterization, Risk, and the Public

This chapter provides the results of the investigations conducted for site characterization, groundwater modeling, risk assessment and risk management, and management of public interaction at the Massachusetts Military Reservation. The results are divided into four sections.

3.1 Site Characterization of the MMR LF-1 Plume

Site characterization investigations followed two main topics with respect to this report. The first involved describing the nature and extent of the chemical contamination in the groundwater. The second involved analyzing tests for hydraulic conductivity to determine parameters that could be used for modeling contaminant migration.

3.1.1 Groundwater Contamination

As part of the Superfund Remedial Investigation process, 73 wells at different locations and different depths were tested for 34 of the most likely compounds. The EPA standard for drinking water sets individual maximum contamination levels (MCLs) for most of these compounds. 28 out of the 73 wells had at least one contaminant which exceeded the MCL. 7 out of the 34 possible contaminants were at levels which exceeded the MCL. These contaminants are vinyl chloride (VC), carbon tetrachloride (CT), trichloroethene (TCE), tetrachloroethene (PCE), 1,4 dichlorobenzene (1,4 DCB), benzene (B), and chloroform (CF). All of these compounds have an MCL of 5 ppb, except for vinyl chloride which has an MCL of 2 ppb. The highest total of all 7 of these contaminants at any one well was 162 ppb.

The highest total of all contaminants sampled at any one well was 236 ppb. (Some of these contaminants have an MCL much higher than 5 ppb.) The highest three individual contaminant readings were CT at 60 ppb, TCE at 64 ppb, and PCE at 65 ppb. One ppb by volume is equivalent to one drop in 15,000 gallons. 162 ppb is equivalent to about 1/3 ounce per 15,000 gallons. At 60 gallons per day of individual water use, 15,000 gallons are used in 250 days. At 236 ppb, the highest total concentration sampled, this works out to about 1 drop of exposure per person per day. The risk assessment section of this report discusses the danger to humans from possible exposure.

Looking at two dimensional log-linear contours of the contamination data points and vertical section filtered contours (see Figures 3-1 and 3-2), a very rough estimate of the total volume of contamination can be made. This is estimated to be about 160 cubic feet or 22 - 55 gallon drums. This mass is distributed over approximately 4.5 square miles. The area where any single MCL level is exceeded is about 2 square miles.

Contamination contours show that little degradation of PCE is occurring. TCE is the degraded product of PCE. The contours show the center of PCE concentration to be downgradient from the center of TCE concentration, therefore the TCE could not be the result of PCE degradation. Instead, this indicates that TCE must be one of the originally dumped contaminants.

Figure 3-1. Contours of Total Contamination Concentration, 20 to 100 PPB

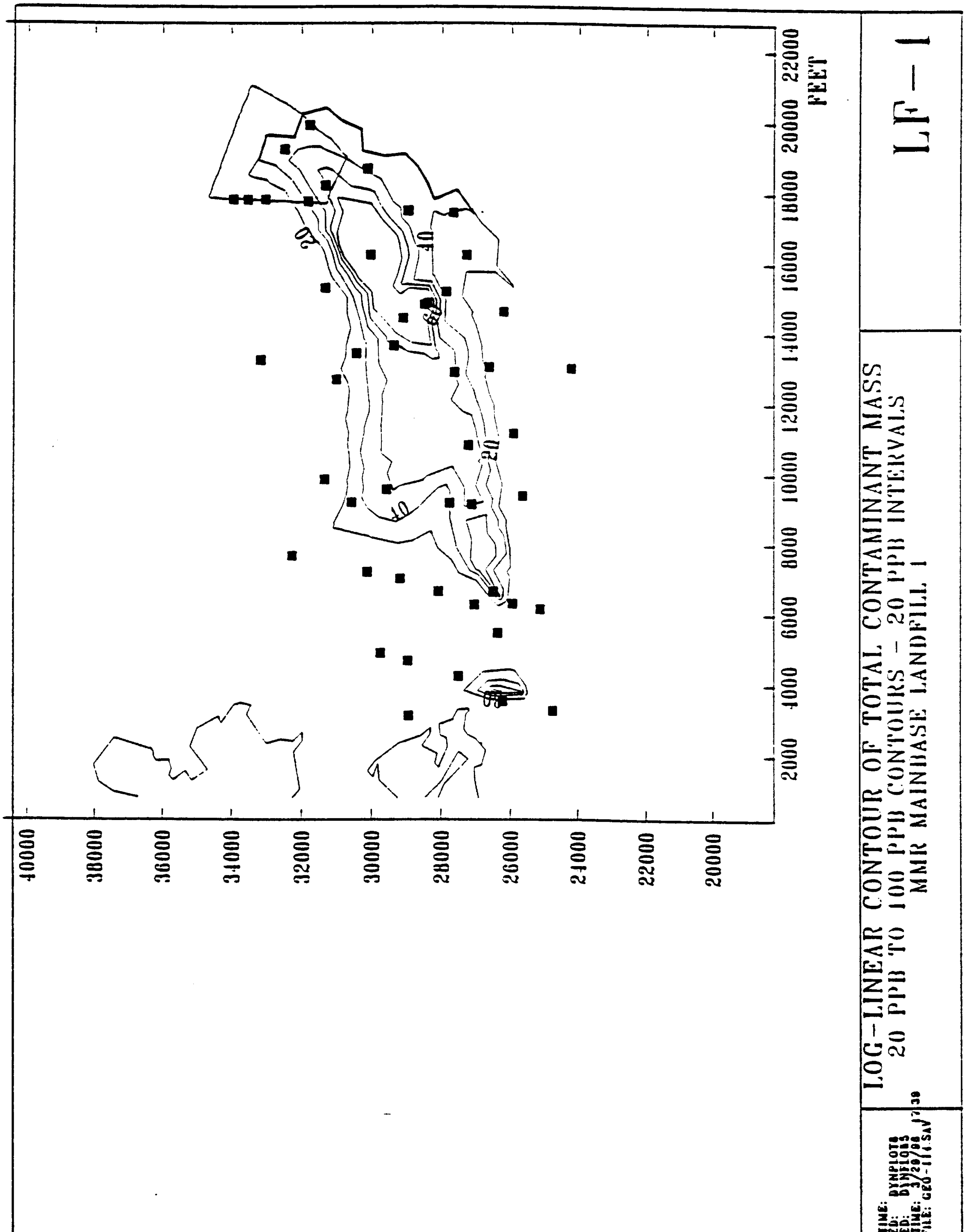
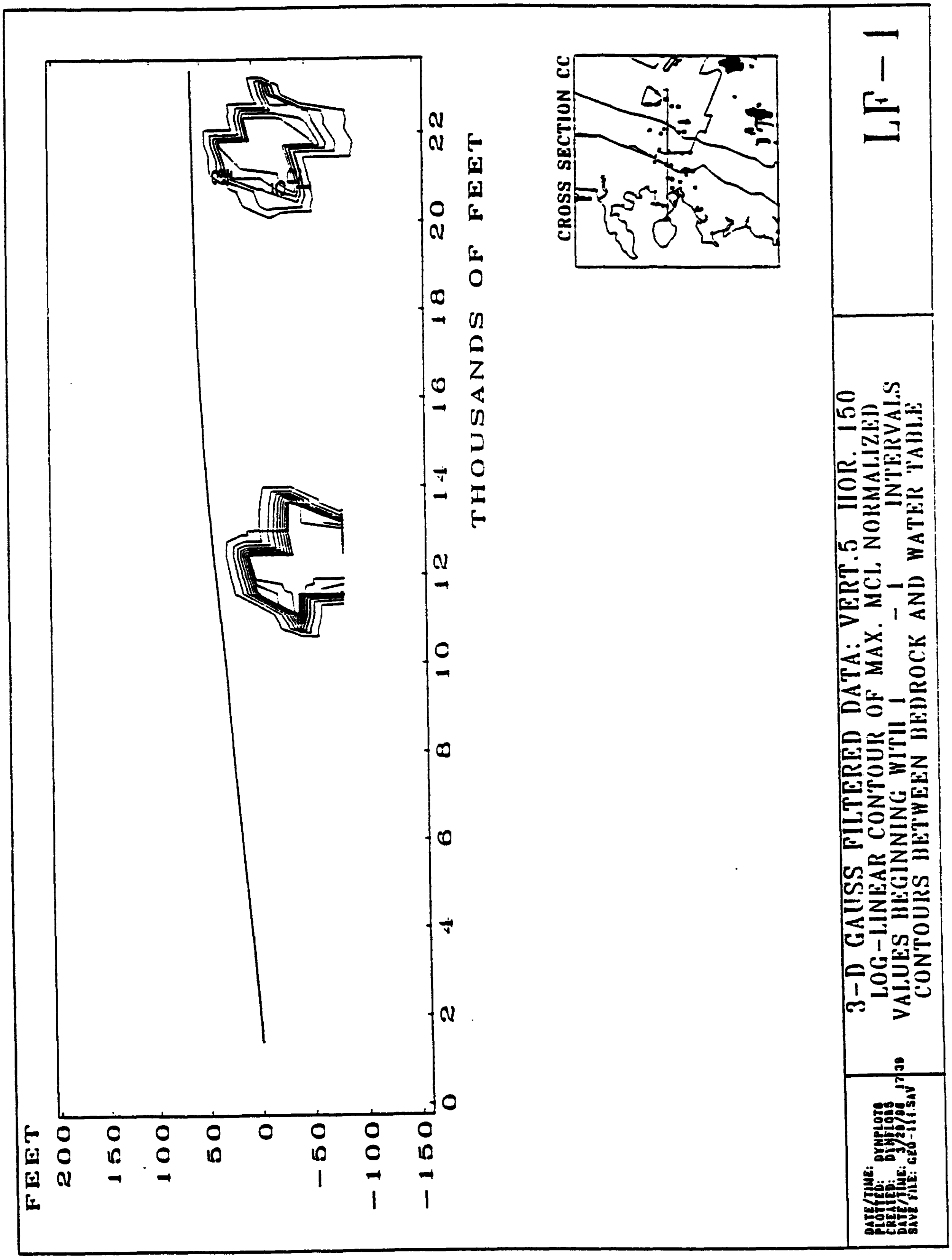


Figure 3-2. Vertical Cross Section of MCL-Normalized Contamination



A comparison can be made between possible contaminant discharge to the ocean through groundwater migration versus the same discharge through a pipe from a hypothetical industrial source. If the contaminant front is considered to be 50 feet thick by 5000 feet wide and moving at a rate of 1 foot per day, this equates to an outfall pipe 2 feet in diameter with a flow rate of 1 foot per second. (A fast walk is about 5 feet per second.) In addition to drinking water standards, the EPA publishes guidelines for allowable contaminant marine discharge beyond the mean low water mark. These standards are considerably higher than those for drinking water. If the landfill plume were being discharged from a single pipe, the EPA would have to decide whether to permit such a discharge. From the given guideline values, and the known contamination levels, it is difficult to say whether a permit would be granted. However, the discharge is, in effect, put through a diffuser over an area 2500 times as large as the hypothetical pipe.

Examining cross sectional contours of contamination (see Figure 3-2), it is seen that a contamination level exceeding the MCL comes within 10 feet of the top of the aquifer. It is estimated that the withdrawal depth of a hypothetical private well pulling 1000 gallons per day to be 13 feet, given a conservative figure for hydraulic conductivity (50 ft/day) and hydraulic gradient (1/100). Therefore, it is possible that private wells located directly over the uppermost levels of contamination could draw in water exceeding the MCL levels for drinking water.

3.1.2 Hydraulic Conductivity

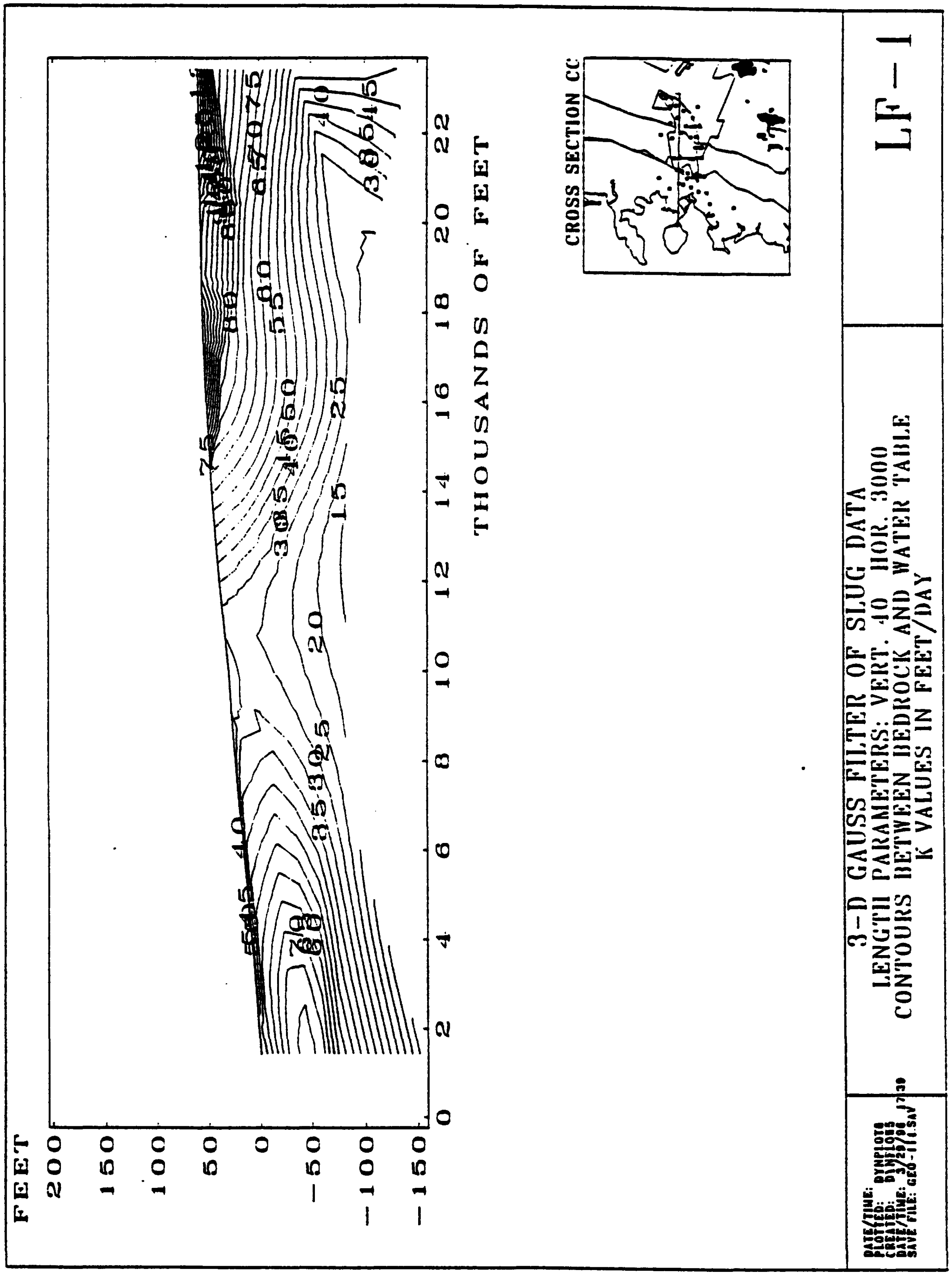
Hydraulic conductivity (K) was determined using 140 grain size samples from 21 well locations and 79 slug test well locations. A comparison of values from these two different tests generally shows very poor correlation. However, a good correlation was seen between the Alyamani/Sen (Alyamani, et al, 1993) and Bedinger (Bradbury, et al, 1990) grain size methods. This is due to the fact that both depend on the grain size fraction d_{50}. Both grain size and slug test data were put through a 3-D gauss filtering process. The resulting data and corresponding contours exhibit a significant correlation between the Hazen and slug methods. However, the Hazen values are much lower.

The filtered slug contours match the general geology of the area, showing a decline in conductivity from north to south and with depth. In addition, the Buzzard's Bay Moraine is clearly seen (see Figure 3-3). The contours also point out a zone of lower conductivity in a region where the contaminant plume appears to be dividing. This finding may provide part of the explanation for the observed migration path. The arithmetic mean of the unfiltered slug test data was 75 feet/day, ranging from less than 1 ft/day to 316 feet/day. The calculated horizontal conductivity from the filtered slug test data had a mean of 85 feet/day and a maximum of 272 feet/day. In addition to hydraulic conductivity, a determination of overall hydraulic anisotropy was made using the filtered slug K values. The number was approximately 3.4. It is very similar to the value of 3.2 determined by Springer for the Mashpee Pitted Plain (Springer, 1991).

3.1.3 Summary

In summary, a large area of groundwater has been contaminated by the MMR Mainbase Landfill 1 with halogenated volatile organic compounds. The contaminant plume is heading west through the Buzzards Bay Moraine. Public and private drinking supply wells are in danger of drawing water with concentration levels exceeding EPA drinking water standards. Hydraulic conductivity trends can be ascertained using gaussian filtered slug test data. Values for horizontal and vertical hydraulic conductivity may be calculated from the filtered data. These values may be used to model migration of the plume. The next section describes the groundwater modeling study.

Figure 3-3. Vertical Cross Section Contours of Filtered Slug Data



3.2 Groundwater Modeling and Particle Tracking Simulation

3.2.1 Objectives and Scope

This section of the report describes a three dimensional groundwater model and particle tracking simulation of the portion of the aquifer that is deemed to affect the spatial characteristics and migration pathlines of the LF-1 plume. The DYN System modeling package developed by CDM, Inc., is utilized for this purpose. The goals of the modeling effort are as follows:

I. Develop a steady state flow model for the study area.

II. Track particles released from a continuous source area and observe migration patterns.

III. Determine flushing time and plume migration with source removed.

IV. Determine sensitivity of model results (plume migration) to the Buzzards Bay Moraine and other geologic features and characteristics of the region.

V. Explore the possibility that the deep plume observed in advance of the main plume is caused by a pool of dense leachate from the landfill sinking below the source area.

3.2.2 DYNFLOW, DYNTRACK and DYNPLOT Systems

The groundwater flow system of the Western Cape is modeled with the DYNFLOW groundwater modeling package. DYNFLOW is a FORTRAN based program that simulates three-dimensional flow using a finite element formulation. A distinct advantage of the finite element based model over a finite difference model like MODFLOW is that the former allows the user the flexibility to use variable sized grid elements. Thus, in regions of interest, the user can obtain higher resolution without having to implement the same degree of resolution throughout the model and obtain significant advantages in terms of computational time and complexity.

DYNTRACK simulates three-dimensional contaminant mass transport and uses the same finite element grid, flow field and aquifer properties that were used in and derived from DYNFLOW. DYNTRACK models either single particle tracking or 3-d transport of conservative or first-order decay contaminants with or without adsorption and dispersion.

DYNPLOT is a graphical pre- and post-processor that can create full color displays in plan view or cross-section of observed data, DYN system calculated data and simulated results. DYNPLOT is also capable of generating the finite element grid used by the flow and tracking models.

3.2.3 Study Area and Grid

The roughly triangular study area of the model was chosen to be large enough to ensure that boundary effects did not unduly influence the calculated flow and head values in the area of concern. The study area, approximately 58 square miles in extent, is depicted in Figure 3-5. The northern and eastern boundaries of the model are streamlines (no-flux boundaries). The western part of the grid area is bounded by the ocean. The ocean-aquifer interface is of particular interest because it determines how far out at sea the LF-1 plume will discharge if it is not completely contained.

The grid covering the LF-1 study area was generated in DYNPLOT, with smaller grid elements in the sources area and presently observed plume locations and progressively coarser grid elements moving away from these locations. The study grid is composed of 3156 triangular elements and 1652 nodes. The grid discretizes the vertical dimension of the study area in 8 layers (9 levels). The bottom (1st) level follows the bedrock contours, while the top (9th) level approximates the surface topography.

3.2.4 Model Formulation

3.2.4.1 Assigned Geologic Materials

The geologic structure of the LF-1 study area was represented as depicted in Figures 3-6, 3-7, 3-8 and 3-9. The geographic locations of the material were assigned according to USGS maps of the region. The Mashpee Pitted Plain (MPP) was represented vertically as two material types and two horizontal sections. This was done to accurately represent the upward coarsening and north-south fining that is observed (LeBlanc, 1986) The Buzzards Bay Moraine (BBM) was defined vertically as four different material of increasing permeability upwards and two horizontal divisions. The Buzzards Bay Outwash (BBO) was depicted by two vertical materials, coarsening upwards. All three deposit types were underlain by a layer of Glacio-Lacustrine deposits (GLS) of varying thickness and bedrock.

3.2.4.2 Source

The LF-1 source was represented by six distinct cells within the source area. In the particle tracking simulation, three cells were defined as being non-sources after 1994. This was done to simulate a successful capping of part of the landfill in 1994 by the IRP.

3.2.4.3 Ponds

Ponds were modeled as a layer of material that was almost infinitely permeable horizontally and with a high vertical conductivity of the order of 500 ft/day. The pond material layer was extended to the observed depth of the each pond. These pond nodes were then assigned a rising head boundary condition. With this method, the material defined as the pond displays a consistent horizontal head and acts as a sink for groundwater upgradient of the pond and a source of groundwater to sections of the grid downgradient. This formulation was considered to most closely approximate the behavior of ponds in the Cape Cod region.

3.2.4.4 Hydraulic Properties

3.2.4.4.1 Hydraulic Conductivity

Estimates of hydraulic conductivity for the LF-1 region have been made through field investigations. Many slug tests, and laboratory tests of soil samples have been carried out for the sediments found in the Cape Cod region. The previous section on site characterization carries a full discussion of these empirical findings. For the purposes of the groundwater model, hydraulic conductivities proved to be the parameter to which the flow model was most sensitive. Hydraulic conductivity values of each sediment type were considered a variable input, and were assigned values within an empirically determined range obtained from literature in calibrating the flow model. The final values of hydraulic conductivities assigned to each geologic material are included in Table 3-4.

Material	K_x, K_y ft.day	K_z ft/day	Long. Disp ft.	Trans. Disp ft	Disp Ratio vert./horiz
Lacustrine	15	5	90.0	3.3	0.03
Fine Sand West	80	27	90.0	3.3	0.03
Coarse Sand West	180	60	90.0	3.3	0.03
Fine Sand South	135	45	90.0	3.3	0.03
Coarse Sand South	210	70	90.0	3.3	0.03
BBM Low -North	30	10	90.0	3.3	0.03
BBM Med Low-North	110	33	90.0	3.3	0.03
BBM Med High-North	150	50	90.0	3.3	0.03
BBM High-North	170	57	90.0	3.3	0.03
BBM Low -South	15	5	90.0	3.3	0.03
BBM Med Low-South	60	20	90.0	3.3	0.03
BBM Med High-South	100	33	90.0	3.3	0.03
BBM High-South	135	45	90.0	3.3	0.03
Nant. Ice Deposits	190	63	90.0	3.3	0.03
Pond Material	10^{-5}	10	90.0	3.3	0.03
Fine Sand North	140	47	90.0	3.3	0.03
Coarse Sand North	270	90	90.0	3.3	0.03
Fine Lacustrine	10	3	90.0	3.3	0.03

Table 3-4. Hydraulic Conductivities and Dispersivities used in flow and mass transport models.

3.2.4.4.2 Dispersivity

Accurately characterizing the dispersivity at a field site is essential in predicting the transport and spreading of a contaminant plume. Due to natural heterogeneities in the field that cause irregular flow patterns, field-scale dispersivities are several orders of magnitude larger than laboratory scale values (Gelhar et al., 1992). In this model, a tabulation of field-scale dispersivity data is used to obtain suitable values of the dispersivity coefficients while taking into account the scale of the LF-1 source. These values are also included in Table 3-4.

3.2.4.4.3 Effective Porosity

Porosity estimates for the outwash in the LF-1 study area range from less than 1% to over 30% (CDM Federal, 1995). These values are somewhat lower than expected from tracer tests of Cape Cod, which range from 38-42% (Masterson and Barlow, 1994). It was decided to use an effective porosity value of 39% throughout the model.

3.2.4.5 Boundary Conditions

3.2.4.5.1 Saltwater-Freshwater Interface

The saltwater-freshwater interface determines where the landfill plume, if not fully contained, will discharge in to Megansett, Red Brook and Squeteague harbors. The steepness and the distance from shore of the interface depends on the aquifer discharge and geologic characteristics of the coastal region. Available geologic information does not indicate the existence of low permeability layers above the aquifer near the shore that will force the salt-fresh interface further into the ocean. Therefore, for the purposes of this report, it is assumed that the location and shape of the salt-fresh interface along the Western Cape Cod shore are determined entirely by the discharge and hydraulic conductivity of the aquifer. The distance from the shore to the salt-fresh interface was calculated to be approximately 500 ft.

3.2.4.5.2 No-Flux Boundaries

No-flux boundaries are modeled in DYNFLOW by assigning all nodes on streamlines at the edge of the study area a "free head" boundary condition. It is assumed that the no-flux boundaries are far enough from the areas of the model we wish to observe that they do not influence the calculated values of head and velocity.

3.2.4.6 Recharge

Natural recharge is the largest source of replenishment of the West Cape aquifer system. This natural recharge is composed entirely of rainfall infiltrate through the surface layer. Cape Cod on average receives 46 inches of rainfall annually. Nearly half of this precipitation, or 46-50%, infiltrates to the groundwater system through the highly permeable top soil (LeBlanc et al., 1986). There is little or no surface runoff due to the permeable nature of the soils and the small topographic gradients present in this region. Artificial recharge and pumping is considered to be negligible in this region in comparison with the natural recharge.

3.2.5 Results

The calibrated flow model agreed with observed water table measurements at 106 wells within 0.044 ft mean difference and 2.159 ft standard deviation. Figure 3-10 shows the calibrated model results and calculated water table contours. The calculated contours are also consistent with observed water table contours in the region.

The flow model was found to be very sensitive to the difference in permeability between the moraine and surrounding deposits. This sensitivity is highlighted by the curvature of the model calculated head contours, which in turn significantly influence the migration pathlines of a contaminant released at the LF-1 site. The sensitivity of the particle paths to head contours is enhanced by the fact that the LF-1 source area is located close to the point where north south head contours change to an east-west orientation.

The first particles released at the LF-1 site will migrate to the ocean in 50 years. Figure 3-11 shows a 51 year mass transport simulation in plan view, with particles reaching the ocean interface. Figure 3-12 is a cross section of the simulated plume. Thus, assuming that the volatile organic compounds of concern at this site were released in 1945, the predicted extent of the plume reaches the ocean discharge face by 1996. The initial discharge point is at Red Brook Harbor. This finding is in agreement with the Op-Tech Data Gap Report which concludes that the LF-1 plume has now reached Red Brook Harbor (Op-Tech, 1996).

If the entire landfill is successfully capped by the year 2000, and the contaminated groundwater is allowed to flush unmitigated into the ocean, the DYNTRACK simulation time of 110 additional years is required for all LF-1 derived contaminants in the aquifer to travel beyond the Buzzards Bay Moraine. A further 55 years is required for all the contaminant particles to be discharged from the aquifer.

The predicted plume exhibits the same differential North and South Lobe travel times observed in the field. In the model, the presence of a low-permeability layer in the moraine causes the southern part of the plume to be retarded. The northern section, by virtue of having to travel a shorter distance to the moraine, is at a higher elevation than the southern part of the plume and thus travels through a higher permeability layer of the moraine. These differential travel velocities through the moraine cause the distinct northern and southern lobes observed in the simulated plume. Figure 3-13 is a north-south cross-section of the plume at the point of entry into the moraine, showing the differential elevations of the particles from north to south.

The previous finding that the portion of the plume at a lower elevation is retarded by the presence of a lower conductivity layer of moraine deposits indicates that the deep plume observed near the shoreline cannot be simulated by a sinking source of contaminant in this model formulation. A tenable explanation for the observed deep northern plume is that the down-sloping bedrock surface near the shoreline causes the faster moving simulated northern lobe to sink further due to infiltration as it traverses the Buzzards Bay Outwash towards the shoreline. Since the slower moving southern lobe is still in the moraine, the leading edges of the northern lobe near Red Brook Harbor now appear to be a northern plume lobe at a lower elevation.

If an extraction well system is constructed along Route 28, and it is assumed that the extraction pumping and infiltration are carried out so that the hydraulic system is relatively unchanged, the uncaptured section of the LF-1 plume will take a further 12 years to completely discharge into the ocean. This result was obtained assuming that the portion of the plume upgradient of the extraction well fence is fully captured.

In summary, the groundwater flow and particle transport model provides results that are similar to field observations. The Buzzards Bay Moraine exerts a great deal of influence on the regional hydrologic system. The geologic characteristics assigned in the flow model to the BBM defines the shape of the regional head contours and thus the travel path and velocity of the simulated plume. Therefore, it is essential that the geology of this moraine be properly identified if a flow and particle tracking model that can accurately represent the region is to be formulated. In the absence of such data, any groundwater flow model of the LF-1 region will contain a significant degree of uncertainty and error. The models developed in this study can be used to determine the effects of an extraction system to contain or capture the LF-1 plume and also as a means of designing an efficient capture system for this contaminated site. The following section addresses the risks associated with the LF-1 plume and how these risks can be managed.

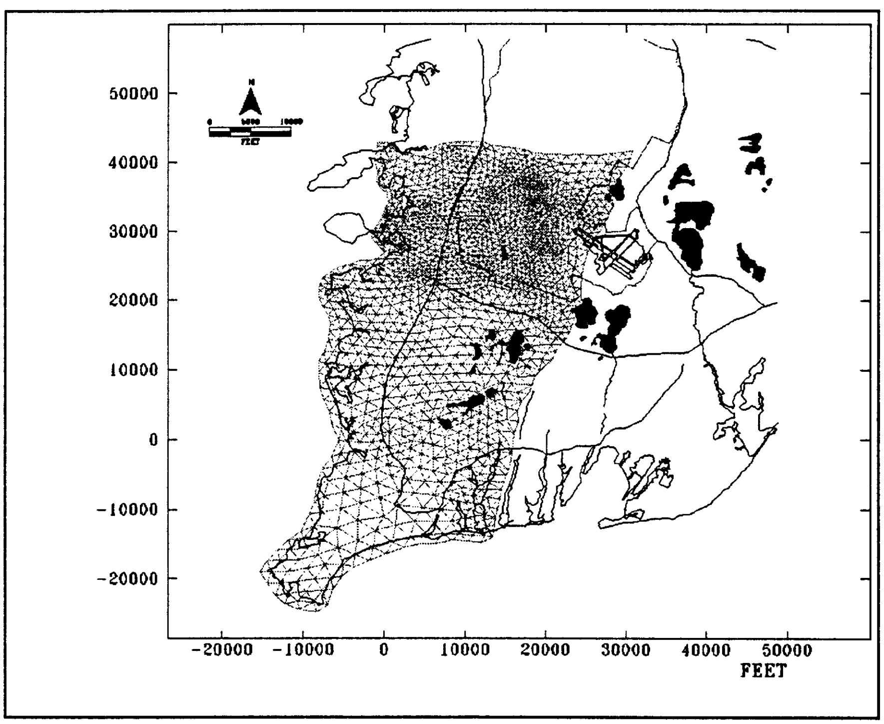


Figure 3-5. LF-1 Study Area and Finite Element Grid

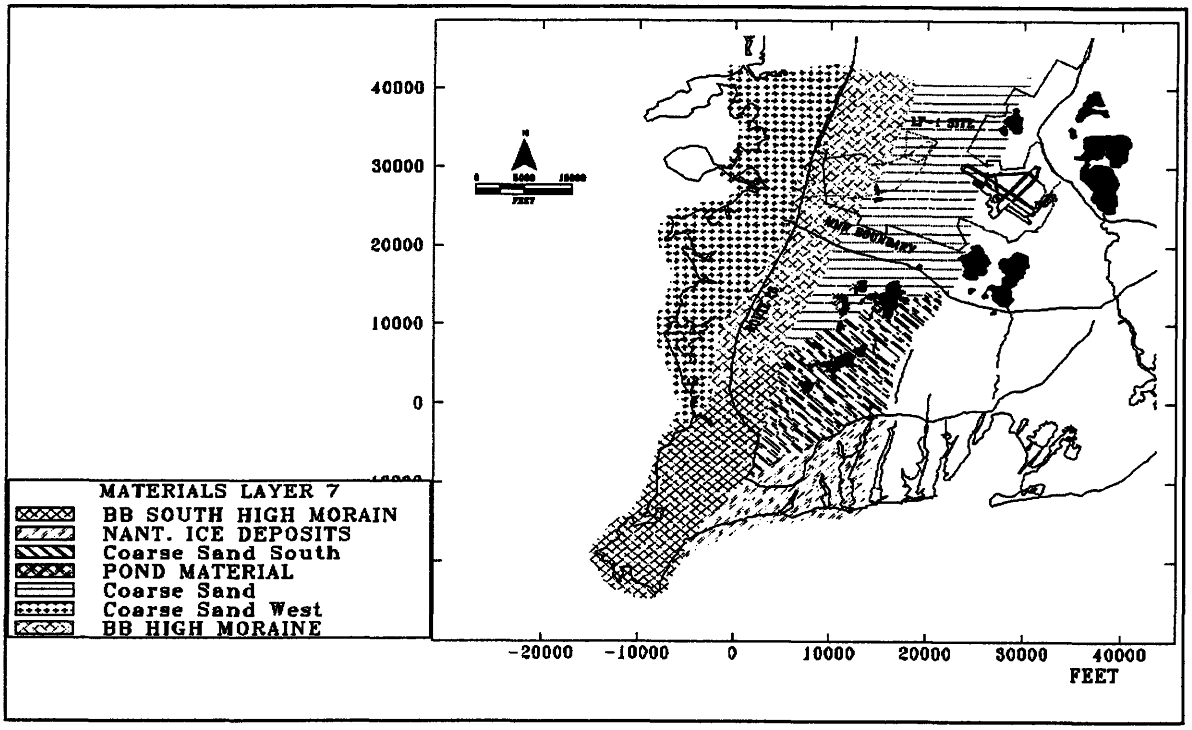


Figure 3-6. Plan View of LF-1 Study Area with Assigned Geologic Materials

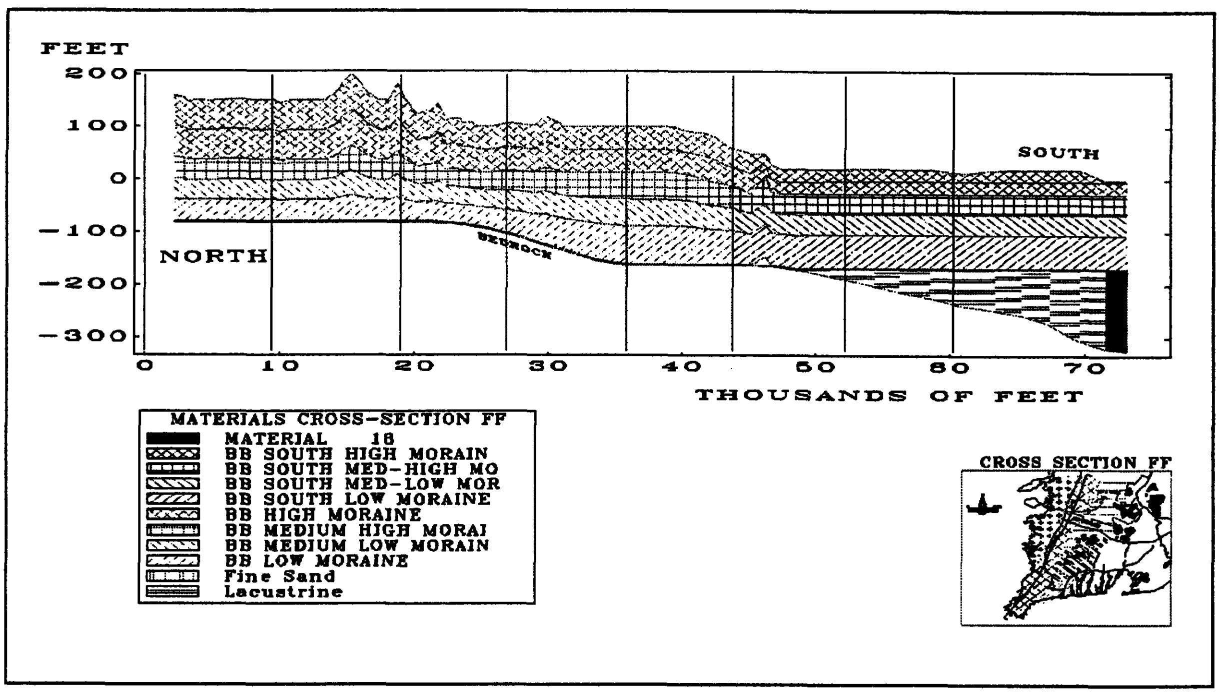


Figure 3-7. Cross-sectional View of Buzzards Bay Moraine Deposits

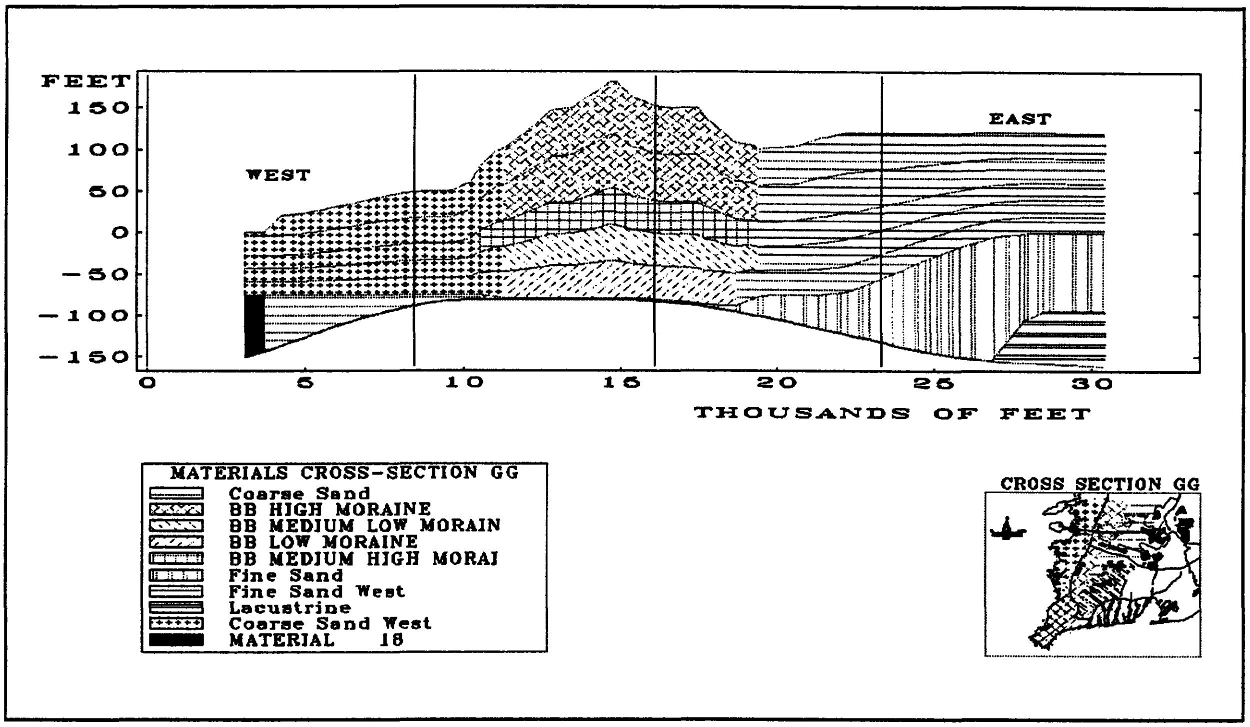


Figure 3-8. East-West Cross-section of Study Area near Buzzards Bay

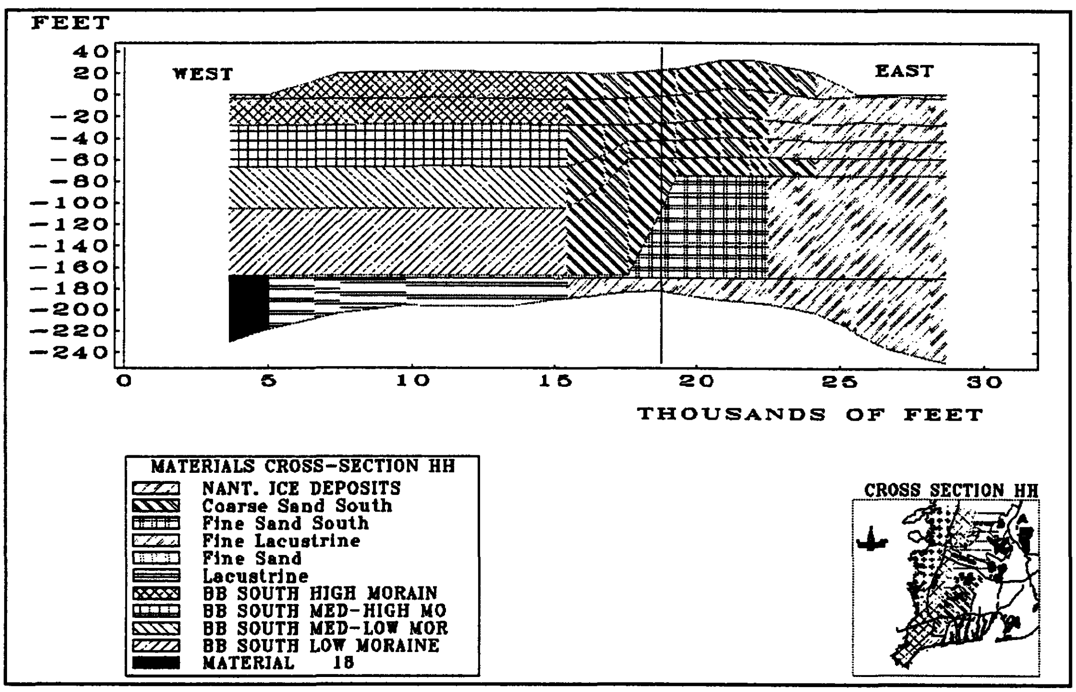


Figure 3-9. East-West Cross-section of Study Area near Nantucket Sound

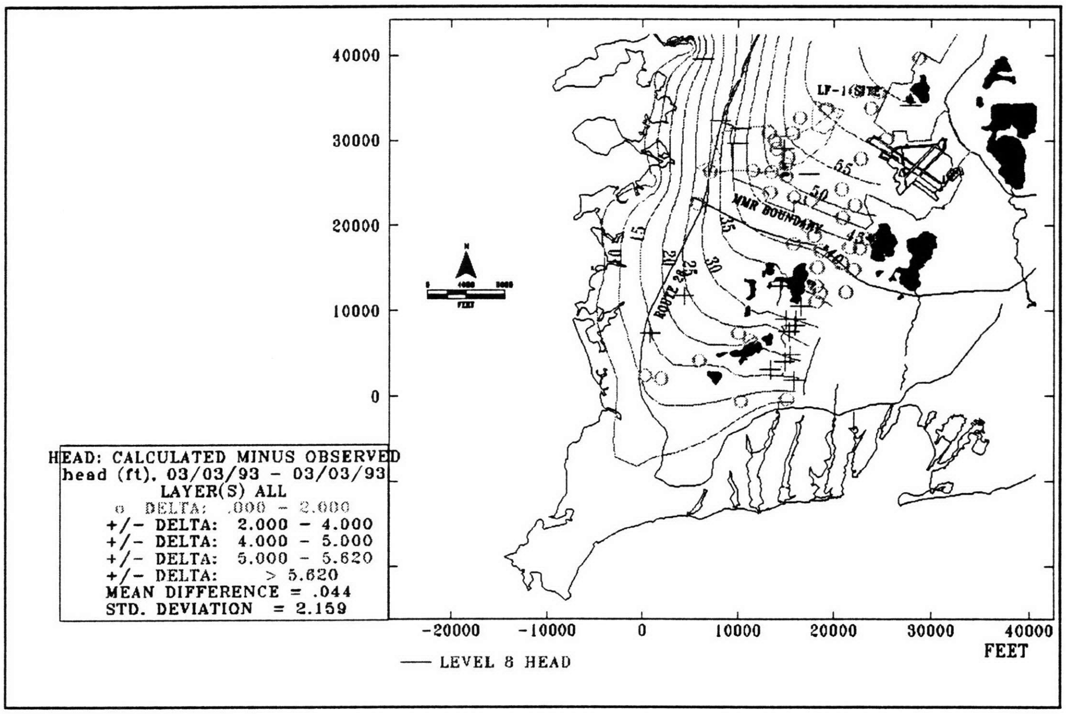


Figure 3-10. Calculated Water Table Elevation Contours and Flow Model Calibration Results

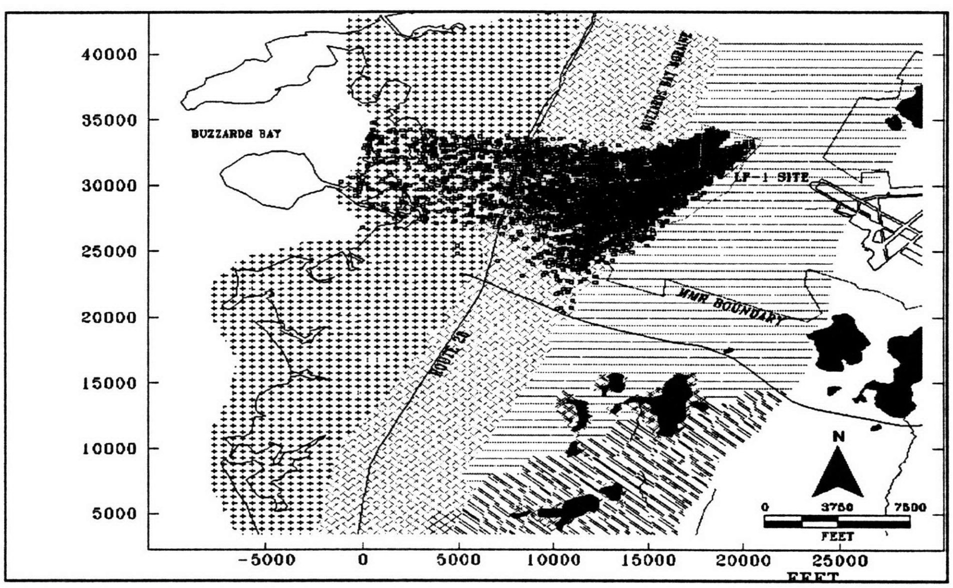


Figure 3-11. Plan View of Simulated LF-1 Plume. Buzzards Bay Moraine is also shown

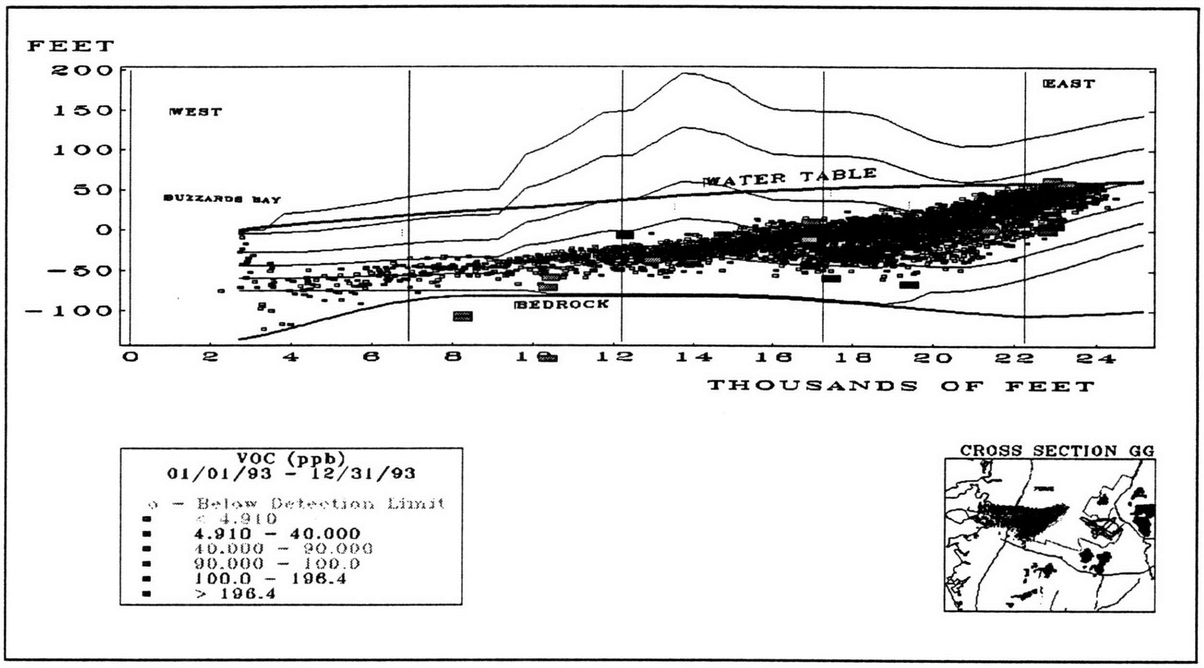


Figure 3-12. Cross-section of Simulated LF-1 Plume and Observed Contamination Locations

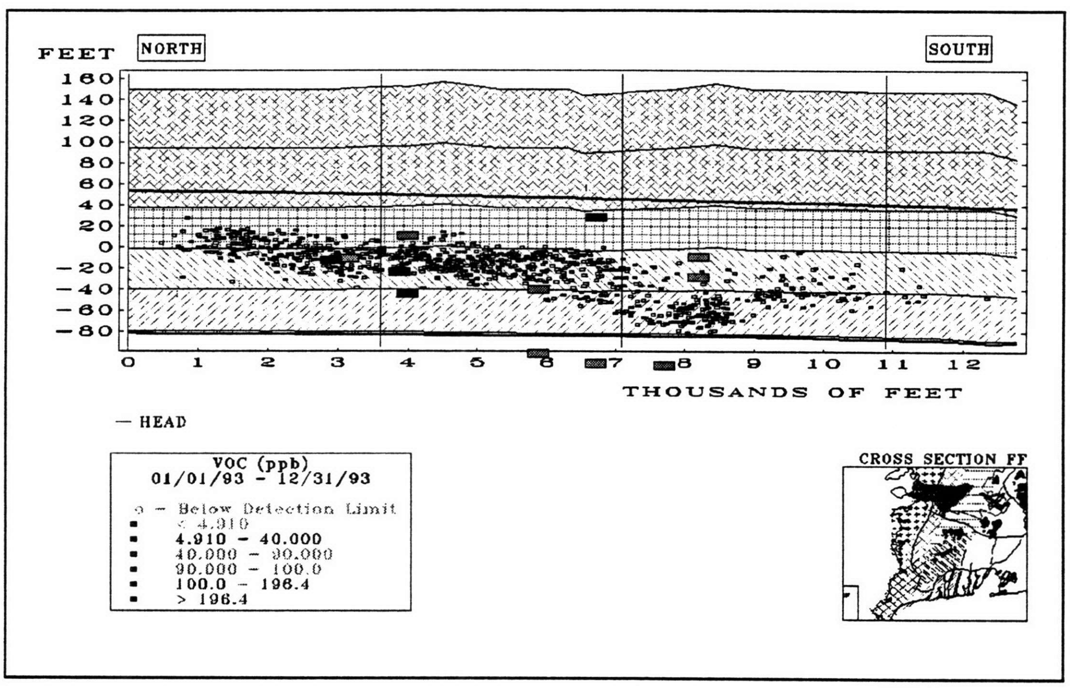


Figure 3-13. Cross-sectional View of LF-1 Plume as it enters the Buzzards Bay Moraine

3.3 Risk Assessment & Management of Risks

The IRP's Remedial Investigation (RI) Report and their Final Risk Assessment Handbook (RAH) present an evaluation of potential adverse effects to human health from materials identified in the MMR LF-1. The MMR site has been classified using EPA guidelines which were not specifically developed for the MMR site. The accuracy of the health and environmental risk scores are limited by the constraints of the EPA's deterministic risk assessment model.

Cancer risk is the statistical increase in mortality rate for a member of the local community who has been exposed to carcinogenic materials identified in the MMR LF-1 as compared to the rate for a member of the local community if the MMR LF-1 did not exist. It is the probability of an event occurring and the magnitude of the effect which an event will likely produce. More simply, cancer risk is the product of the probability of dying from cancer because of exposure to carcinogens and the probability of exposure to carcinogens.

3.3.1 Toxicology

According to the EPA guidelines (cited in both the RAH, 1994 and LaGrega et. al., 1994), toxicology and dose are to be calculated by following specific protocols. In terms of toxicology, carcinogens are considered to vary greatly in their potency. "When considering lifetime cancer risk to humans, it is widely accepted that carcinogenesis works in a manner such that it is possible, however remote, that exposure to a single molecule of a genotoxic carcinogen could result in one of the two mutations necessary to initiate cancer". (LaGrega et. al., 1994, p. 277). Therefore, the calculation of carcinogenic risk from toxicology involves the use of cancer potency factors which are basically the slopes of the dose-response curves for carcinogens which are extrapolated to zero for extremely small doses. These extrapolated slopes are commonly referred to as cancer slope factors (CSFs) and they are used for the toxicological component of the EPA's acceptable risk calculations. CSFs are maintained in the EPA's Integrated Risk Information System (IRIS) database.

Many papers have been published which comment upon the uncertainty of the EPA's CSFs. In addition, "the EPA is well aware of the problems associated with overly conservative risk estimates and has repeatedly stressed that the unit cancer risk estimate only provides a plausible upper limit for a risk that can very well be much lower. The problem is that, in reality, official EPA unit risk estimates are widely used , more or less, as absolute standards." (LaGrega et. al., 1994, p.280). Due to insufficient expertise in toxicology, this report will not offer an opinion concerning specific toxicological uncertainty of the EPA's CSFs.

3.3.2 Dose

In terms of dose calculations, it is important to understand the environmental pathway. Therefore, for this cancer risk evaluation it is important to identify the following:

- carcinogens
- source of carcinogens
- release mechanisms
- transport mechanisms
- transfer mechanisms
- transformation mechanisms
- exposure paths

- exposure point concentrations
- receptors

However, it is interesting to note that in performing an EPA risk assessment, only the carcinogens and the exposure point concentrations are used to calculate risk. Although the other seven above-referenced factors are essential for developing spatially distributed exposure point concentrations, EPA protocol requires maximum detect concentrations for maximum or upper bound risk calculations. In addition, EPA protocol requires arithmetic averaging of detect concentrations for mean risk calculations. That is to say, two sites with hazardous materials at similar concentrations with entirely different hydrogeologic conditions, would have the same risk according to EPA guidelines. However, at their discretion, EPA will review risk assessments which incorporate site-specific conditions into their calculations.

3.3.3 Identification of Hazardous Materials

Hazardous materials are broadly defined as non-carcinogens which are known to have harmful systemic effects upon humans, and carcinogens which have a propensity to initiate and promote cancer. Both terminal and "quality of life" health problems from exposure to hazardous materials are primary human health concerns. Because of these health concerns, human exposure to hazardous materials, especially carcinogens, is a source of risk and is of primary concern for risk assessment and management. However, for this report, only the carcinogenic materials identified in the MMR LF-1 are being evaluated for potential risk; they are identified in the risk spreadsheets presented in volume 2 appendix A.3.

According to Boston University's School of Public Health Upper Cape Cancer Incidence Study which was prepared under contract to the Massachusetts Department of Public Health, cancer incidence rates for the MMR regional area have increased at a relative rate of approximately fifty six (56) percent overall (BUSPH, 1992). In addition, according to the Journal of the American Medical Association cancer incident rates are increasing steadily for the United States at a relative rate of approximately forty four (44) percent overall (JAMA, Vol. 271, No. 6, 1994). Furthermore, it is generally accepted that approximately twenty five (25) percent of all annual deaths in the US are caused by cancer. When the uncertainties presented in the above-referenced reports are taken into account, both the MMR cancer rate and the US cancer rate are very similar. Since these cancer rates are so similar, it is difficult to discern if the cancer rate increase at the MMR region is caused on account of reasons which are linked to the background national cancer rate increase, or if cancer rate increase near the MMR is tied to the release of carcinogenic materials at the MMR site.

3.3.4 Review Existing Reports

Part of this investigation was a comprehensive review of the RI, and the RAH which are relevant to risk assessment for the MMR LF-1. An examination of the methodology used, the consistency of the reports with respect to the EPA's regulatory guidelines, and independent spreadsheet calculations using the equations and numerical values which are cited in the above-referenced reports supplied similar results. This three part process confirmed the consistency of the reporting which has been provided to MIT to calculate risk and formulate risk opinions. Independent spreadsheet calculations are included in volume 2 appendix A.3. As the MMR LF-1 is part of an on-going clean-up, new and updated data from the above-referenced reports has been included, as required, to present the most current EPA approved health risk connected with the MMR LF-1.

3.3.5 Uncertainty

In all statistically intensive calculations there are uncertainties specific to the numerical model which is being used. Since the EPA's model is the requisite regulatory guideline for Superfund sites, their model is the one which is being scrutinized. The EPA's deterministic model does not distribute uncertainty uniformly. When combined, concentration uncertainty and cancer slope factor (CSF) uncertainty account for approximately 97% of total risk uncertainty. Approximately 80% - 95% of the total risk uncertainty is CSF uncertainty. (Hines, J.J. 1996) The EPA understands that their methods are statistically conservative and consequently will tend to overestimate risk, because the EPA incorporates policy constructs into risk quantification calculations. Basically, the EPA uses regulated risk assessment as opposed to probabilistic risk assessment coupled with regulations for risk management. Ultimately, risk regulated by the EPA is as uncertain as the EPA's CSFs. Recently, according to several major journals including the April 17, 1996 issue of the Wall Street Journal, the EPA has proposed policy changes for their assignment of CSFs. This should decrease the statistically localized risk uncertainty inherent within EPA regulated risk assessment calculations.

3.3.6 Assessment of Risk from Ingestion of Contaminated Shellfish

From the current data of the LF-1 plume, the contaminants are projected to discharge into Red Brook and Megansett harbors of Buzzards Bay (OpTech, 1996, CDM Federal, 1995). The shallow tidal flats of these harbors support a rich population of local shellfish species. Soft shell clams, quahogs (hard clams), oysters, bay scallops, surf clams, mussels, and conch are harvested from these harbors by local commercial and recreational fishermen. Since metals are part of the LF-1 plume contaminants and shellfish have been shown to bioaccumulate metals in their body tissue, the potential discharge of the plume into the harbors along the shoreline poses a risk to the coastal marine shellfish populations as well as to consumers of shellfish.

	Max. C[@,1] (ug/l)	Max. C[@,2] (ug/l)	Oral SF	Oral RfD	Cancer Risk[1]	Cancer Risk[2]	Hazard Index[1]	Hazard Index[2]
Aluminum	20900	10200	NA	1	NA	NA	3.18217	1.55302
Antimony		2.6	NA	0.0004	NA	NA	0	0.98967
Arsenic	3.5	8.4	1.75	0.0003	0.00093	0.00224	1.77633	4.2632
Barium	400	107	NA	0.07	NA	NA	0.87004	0.23274
Beryllium	3.6	1.1	4.3	0.005	0.00236	0.00072	0.10963	0.0335
Cadmium	2	2	NA	0.001	NA	NA	0.30451	0.30451
Chromium#	54.2	66.3	NA	0.005	NA	NA	1.65047	2.01893
Copper	48.7	28.2	NA	NA	NA	NA	NA	NA
Cyanide	16.4		NA	0.02	NA	NA	0.12485	0
Iron	134,000	24000	NA	0.5	NA	NA	40.8049	7.30834
Lead	27.8	9.8	NA	NA	NA	NA	NA	NA
Manganese	5040	824	NA	0.14	NA	NA	5.48126	0.89614
Mercury	0.3*	0.3*	NA	0.0003	NA	NA	0.15226	0.15226
Nickel	24.4	184	NA	0.02	NA	NA	0.18575	1.40077
Vanadium	33	41	NA	0.007	NA	NA	0.71778	0.89179
Zinc	262	184	NA	0.3	NA	NA	0.13297	0.09338

Notes:
1 Derived from CDM Federal (1995)
2 Derived from OpTech (1996)
@ Maximum total concentration
# Chromium (VI) values are used
* Maximum dissolved concentration

SF = Cancer slope factor
RfD = Non-cancer reference dose
NA = Not available

Table 3-14. Maximum Cancer and Non-cancer Risk for Each Metal

The results of maximum cancer and non-cancer risk assessment of consuming contaminated quahogs over a life time are calculated for each metal in Table 3-14. The maximum concentration of metals detected in well samples from the LF-1 plume are derived from the reports of CDM Federal (1995) and OpTech (1996). The oral cancer slope factors (SF) and non-cancer reference doses (RfD) of the metals are obtained from the *Risk Assessment Handbook* for the MMR published by Automated Sciences Group (1994). Using the CDM Federal (1995) data, the maximum cancer risk from consumption of tainted quahogs is 3.3E-03. This risk is interpreted as the incremental increase in probability of developing cancer above background level for each exposed resident. The United States Environmental Protection Agency (USEPA) acceptable risk standard ranges from 1.0E-06 to 1.0E-04. The standard is set independently for each site and case. The increased risk of 3.3E-03 for each exposed resident is above the highest acceptable USEPA standard. A maximum cancer risk of 3.0E-03 is calculated when maximum concentration of metals from OpTech (1996) data is used in the assessment. The cancer risk for humans from consumption of tainted quahogs are derived from only two metals - arsenic and beryllium - since these are the only metals with published cancer slope factors.

The overall maximum hazard index (HI) for non-cancer risk from potential exposure to the contaminated quahogs are 55.5 and 19.1, when CDM Federal (1995) and OpTech (1996) data, respectively, are used in the assessment. The USEPA's acceptable HI standard for non-cancer risk is 1.0. Calculated HI that are above the USEPA standard poses possible non-cancer deleterious health effects to the exposed population. The total maximum cancer and non-cancer risks from contaminated quahogs are summarized in Table 3-15.

	Maximum Cancer Risk	**Maximum Hazard Index**
CDM Federal Data	3.3E-03	55.5
OpTech Data	3.0E-03	19.1

Table 3-15. Total Maximum and Non-cancer Risks from Consumption of Tainted Quahogs

The risk assessment results show that both cancer and non-cancer risks are above the USEPA standards. The USEPA risk standards are set at levels that adequately protect human health and the natural environment. The calculated risk results indicate that tainted quahogs from the coastal harbors where LF-1 plume is predicted to discharge pose significant risk to consumers of shellfish from these harbors. The calculated risk estimations are based on worst case assumptions. Thus, the risk is a conservative estimate and indicates the maximum risk posed to human health. The methodology and assumptions used in the current risk calculations are detailed in Appendix A4 (Lee, 1996). From these results, it is recommended that a monitoring program for shellfish harvested from Red Brook and Megansett harbors be implemented.

3.3.7 Qualitative Assessment of Potential Ecological Risk

Since quahog clams are predicted to bioaccumulate metals, the discharge of the LF-1 groundwater plume into Red Brook and Megansett harbors will have detrimental effects to the coastal ecological system. Quahogs are a primary food source for certain marine species that reside in the coastal harbors of Buzzards Bay. The contamination of the quahog clams will reduce its population thus triggering a decline in the populations of marine species that depend on quahogs as their sole food source. The decline in population of key species in the ecosystem will lead to an overall decline of the entire ecosystem.

The bioaccumulation of metals by quahog clams can also have detrimental effects on the ecosystem through another mechanism. Since quahog clams do not reside at the top of the coastal ecosystem food web, they are consumed by higher order species. In this process of nutrient transfer up the food chain, contaminants accumulated within lower food chain organisms are also transferred up the food web. Thus, tainted quahogs can potentially transfer accumulated metals to higher order food chain species. The bioaccumulation of metals in higher order organisms will also lead to the decline of a particular population of species and the ecosystem as a whole.

3.4 Public Perception: Management of Public Interaction at the MMR

An analysis of the approaches used to manage public interaction at the Massachusetts Military Reservation was undertaken to characterize the evolution of public perception of risk posed by past activities at the MMR. Public meetings at the MMR between January 15 and March 31, 1996, were attended. In addition, a comparison of management approaches at other bases was carried out. This included interviewing personnel at military bases in California and Arizona. As part of the analysis, suggestions future approaches at IRPs were explored. This included the design of public opinion surveys to be carried out early in the IRP process. Other suggestions for future approaches are also presented.

3.4.1 Public Perception in Superfund Cleanup

In any scenario where pollution is an issue, there is frequently a gap between the perceived risk to human health and the actual risk posed by contamination. Because of scientific uncertainty in risk assessment, often times, the actual risks are not known, and so the perceived level of risk results from speculation by many parties. In the siting of hazardous waste facilities, the potential threat to human health results in the NIMBY ("Not in my backyard") syndrome. Often times this "potential threat" is a perceived one. Public interest groups have fought many a facility siting and won, not due to actual risk, but because of a perceived one. In Superfund cases, unlike potential hazardous waste facility sitings, contamination has already occurred, but there is still a question of whether the contamination poses a real threat to public health. The gap between actual and perceived risks in this case results in the answer to the question of "how clean is clean?" becoming a policy, rather than a scientific, one. Groundwater contamination at the Massachusetts Military Reservation Superfund site is perceived to be a problem, and steps are being taken to remediate this problem to the greatest extent feasible. Public opinion has defined "the greatest extent feasible" as the level to which groundwater is treated to "non-detect" levels for contaminants that pose threats to human health. In private sector cases, economics would figure into the calculation of feasibility of cleanup, but in the case of the MMR, where an entity as large as the federal government is funding the cleanup, the public believes that "anything is affordable" and therefore feasible.

3.4.2 History of Public Involvement at the MMR

The initial approach to management of public interaction surrounding the Installation Restoration at the MMR was similar to the "compliance-based" approach many companies take towards environmental regulation--the National Guard Bureau met only the minimum requirements necessary. Actions taken by the NGB were reactive rather than proactive. The NGB promulgated press releases and sent reports to local libraries, as well as holding news conferences after technical meetings, but any actions beyond that were minimal. Technical meetings concerning IRP activities were closed to the public and media, and virtually no public information meetings were held.

During 1990 and 1991, there was a modest effort to increase public involvement in the cleanup at Otis, as the IRP office at the MMR was created to manage the program locally rather than from far away. The "Joint Public Involvement Community Relations Plan" was presented, bi-monthly public information meetings were initiated, site tours/briefings were made possible, a site mailing list was created, and the IRP office began to print quarterly fact sheets that described the IRP activities. Although these fact sheets were limited in scope, they, along with the public information meetings, represented the first real effort to inform the public about specific activities associated with the IRP.

Late-1991 marked a major change in the way public interaction was managed at the MMR. The IRP office began updating technical reports much more frequently, and progress reports were made available to all interested parties. The local IRP office began educating the public by participating on local radio/cable TV programs as well as taking part in neighborhood association meetings. An educational display was created for to be used at these meetings and at libraries, and detailed bi-monthly fact sheets were developed. In addition, all technical meetings were opened to the public and media.

The post-1991 period also has included the creation of many committees that assist the cleanup activities at the MMR. These committees, called "process action teams", are made up of personnel from the MMR, the relevant regulatory agencies, and the public. These process action teams (or "PATs") report to the senior management board, which was created to oversee the restoration. Presently, a total of 8 community working groups hold regular meetings (Karson, 1995). Although the public is highly involved in the IRP process at this point, how much influence the public actually has in the decisionmaking process is still a question.

3.4.3 Design Of Future Approaches At The MMR And Elsewhere

There are several things that should be considered before an Installation Restoration Program is initiated at a particular base or military reservation. Not the least of these is the management of public interaction surrounding the restoration. Public and public interest group opinion are very likely to polarize as soon as contamination and threat to public health are made known. Public distrust of government, especially on the federal level, compounds the fear that public health is in danger and contributes to the belief that any cleanup activities will be inadequate to alleviate the problem of contamination.

There are steps that can be taken to minimize the potential for adversarial relationships developing between all interested parties in base cleanup. Since the public has been involved in the restoration process at the MMR, the relationships between all interested parties have become less of a barrier to cleanup as all parties are seen to have input into the process. However, analysis of the approach used to manage public interaction at the MMR shows that, even though outwardly it appears that all the "right" approaches were taken, public concern is still an issue. This is due to the fact that early on in the MMR

IRP process, the public was not included and was seen more as a "problem" than a potential source of solutions.

4. Remedial Approaches

4.1 Source Containment

4.1.1 Introduction

As part of remediation operations at MMR, several of the cells at the Main Base Landfill have recently been secured with a final cover system. These cells include the 1970 cell, the post-1970 cell, and the kettle hole. The remaining cells (1947, 1951, and 1957) have collectively been termed the Northwest Operable Unit (NOU). Remedial investigation as to the necessity of a final closure system for these cells is ongoing. This proposal is focused on the design of a final closure system for the 1951 cell. The landfill final closure requirements of the Resource Conservation and Recovery Act (RCRA) and Massachusetts Solid Waste Management Regulations will be examined and adapted to site specific conditions. Material and design options for the components of the cover system will be examined and choices made according to performance, availability, and relative cost, as applicable to site-specific conditions. A cross-section of the proposed cover system is provided in Figure 4-1.

4.1.2 Regulatory Review

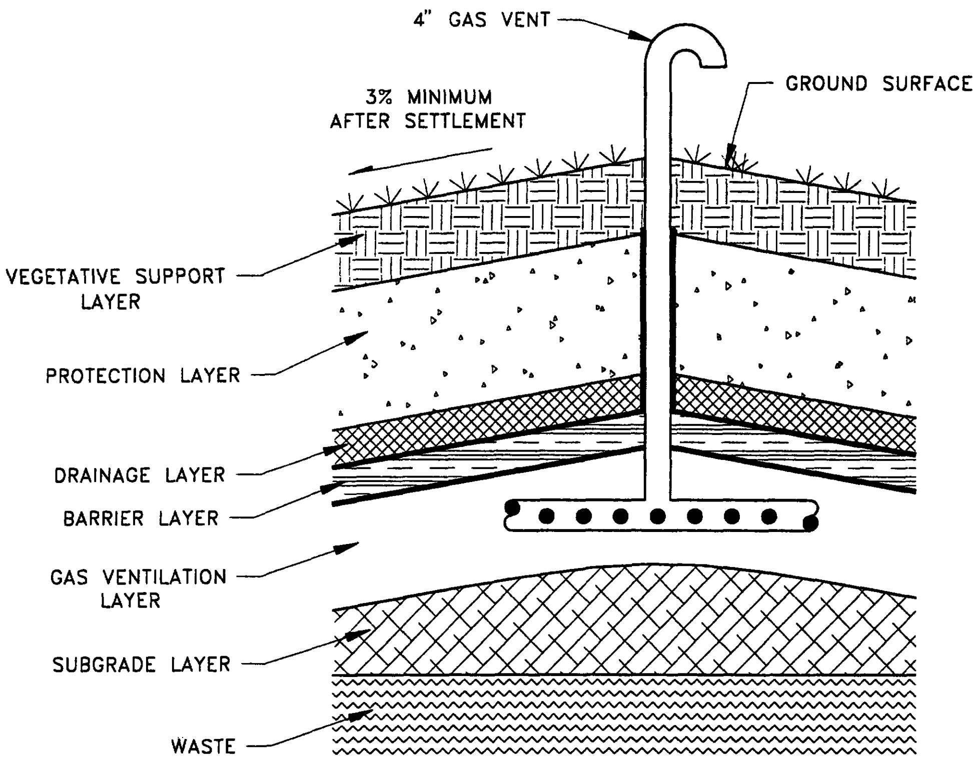
Massachusetts Solid Waste Management regulations specify the following as minimum design requirements for a landfill final closure system (MA DEP, 1993):

- Subgrade layer
- Venting layer with minimum hydraulic conductivity of $1X10^{-3}$ cm/sec
- Low conductivity layer with minimum thickness of 18 inches and maximum hydraulic conductivity of $1x10^{-7}$ cm/sec, or an approved flexible membrane liner (geomembrane)
- Drainage layer with minimum thickness of 6 inches and minimum hydraulic conductivity of $1x10^{-3}$ cm/sec, or a synthetic drainage net (geonet)
- Combined vegetative support / protection layer of minimum thickness 18 inches, with at least 12 inches of soil capable of supporting vegetation.

Subparts G, K, and N of the Resource Conservation and Recovery Act (RCRA) Subtitle C (Hazardous Waste Management) regulations dictate the requirements for hazardous and mixed waste landfill cover systems (US EPA, 1991). The EPA recommends that a final cover system consist of the following (US EPA, 1991):

- A low hydraulic conductivity geomembrane / soil layer consisting of a 24 inch layer of compacted natural or amended soil with a hydraulic conductivity of $1x10^{-7}$ cm/sec in intimate contact with a geomembrane liner of minimum thickness 0.5 mm (20 mil).
- A drainage layer of 12 inch minimum thickness having a minimum hydraulic conductivity of $1x10^{-2}$ cm/sec, or a geosynthetic material of equal transmissivity.
- A top vegetative support / soil layer consisting of a top layer with vegetation or an armored surface, and a minimum of 24 inches of soil graded at a slope between 3 and 5 %.

The EPA does encourage design innovation, and will accept an alternative design upon a showing of equivalency.



LAYER	MATERIAL	SPECIFICATIONS
Vegetative Support	Loam	6" thickness
Protection	Borrow Soil	18" thickness
Drainage	Geonet	Transmissivity>3×10^{-5} m^2/sec
Barrier	Geomembrane over Geosynthetic Clay Liner	Geomembrane: 60mil VLDPE Geosynthetic Clay Liner: Gundseal with 40mil VLDPE Substrate
Gas Ventilation	Screened Borrow Soil	12" thickness Hydraulic conductivity>1×10^{-3} cm/sec

NOT TO SCALE

<u>**Figure 4-1. Cross-section of Proposed Cover Design**</u>

4.1.3 Subgrade Layer

The subgrade layer acts as a foundation for the overlying layers of the cap, and it is also used as a contouring layer to create the appropriate final slope of the cover system. It is recommended that the foundation layer be placed to provide a final grade (after settlement) no greater than 5% and no less than 3%. This slope range provides sufficient grade to promote some surface water runoff while not being so steep as to produce erosion of the surficial soils. Allowance must be made for waste settlement that will occur as a result of the vertical stresses imposed by the weight of the cover materials.

Materials typically utilized for foundation layers include a variety of soils, and some acceptable wastes. At sites such as MMR where soil borrow volumes are relatively plentiful, soil is the obvious choice for the foundation layer. Results of on-site borrow characterization tests (ABB, 1993) have revealed that this material is acceptable for use in the foundation layer. The material is classified as a fine-to-medium sand with trace-to-some fine-to coarse gravel (ABB, 1993). This material has a relatively low fines content and has acceptable compressibility characteristics, therefore it is recommended for use in this layer. The subgrade should be placed in lifts of approximately 8 inches and compacted by 4 to 6 passes of a typical sheepsfoot roller. This placement procedure should result in compaction to approximately 90% of the maximum dry density.

4.1.4 Gas Ventilation Layer

The gas venting layer is a permeable layer containing piping for the collection and venting or recovery of gases produced from waste degradation. Based on the cell composition (predominantly burn-fill), the moist, aerobic conditions provided by the intermediate cover, and the time since placement (over 40 years) it is concluded that gas generation rates at the 1951 cell will be low. Consequently, a passive gas venting system is recommended. It is recommended that material from the "lower layer" of the borrow area be utilized for the ventilation layer. The soil must be screened on a 3/8 inch sieve prior to placement, and then placed with a light machine in a single lift with no further compaction efforts. To collect the gas, PVC collector pipe is bedded in the sand and run laterally along the slope. To vent the gas to atmosphere, it is recommended that a total of ten ventilation risers be installed and spaced equidistantly. Flexible (to accommodate loading and settlement) 4 inch perforated PVC is recommended for the collector pipe, and 4 inch non-perforated rigid PVC is recommended for the risers.

4.1.5 Hydraulic Barrier Layer

The barrier layer is designed to minimize the percolation of water through the cover system directly by impeding infiltration and indirectly by promoting storage and drainage of water in the overlying layers and eventual removal of water by runoff, evapotranspiration, and internal storage (Geosyntec, 1994). This design proposal recommends a composite geomembrane over geonsynthetic clay liner (GCL) as the hydraulic barrier layer. The specified geomembrane is a 60 mil (1.5 mm) textured very low density polyethylene (VLDPE), and the specified GCL is a Gundseal® GCL with a 40 mil (1.0 mm) textured VLDPE substrate placed bentonite-side up.

4.1.6 Drainage Layer

The drainage layer functions to remove water which infiltrates the vegetative support/protection layer. It should be designed to minimize the standing head and residence time of water on the barrier layer in order to minimize leachate production (US EPA, 1989). The recommended drainage layer for this design is an extruded solid rib geonet with factory bonded nonwoven, heat-bonded geotextile on both faces. The composite drainage layer must have a minimum transmissivity of $3x10^{-5}$ m^2/sec.

4.1.7 Surface Layer

The top layer of the cover system is actually comprised of two separate layers; the lower layer termed the protection layer and the upper layer termed the surface layer. On-site or local soil is the most commonly used and typically the most suitable material for the protection layer. Suitable on-site materials are available for use in the protection layer. The on-site borrow materials have been classified as a fine-to-medium sand with trace-to-some fine-to coarse gravel (ABB, 1993). This material has a relatively low fines content and a low organic content, therefore it is acceptable for use in the protection layer. The borrow material should be placed to a thickness of 18 inches using a small dozer with low ground-pressure to protect the underlying cover components. Compaction beyond that which occurs during placement is not necessary.

Vegetation is specified as the surface layer cover, consequently the surface layer will be designed for vegetative support. The on-site borrow material is not well suited to supporting vegetation, therefore it is recommended that loam be imported from an offbase supplier and placed to a thickness of 6 inches. A warm season grass mix is specified as the vegetative cover. Periodic mowing and inspection of the vegetative cover are recommended as part of the Postclosure Program.

4.1.8 Conclusions

It is concluded that this cover system, if constructed with appropriate construction quality assurance / quality control, will satisfy the primary objective of containing the source of pollution, thus minimizing further contamination of groundwater by the waste fill. The composite geomembrane / geosynthetic clay liner barrier layer is theoretically nearly impermeable. Estimates of the hydraulic conductivity of VLDPE geomembranes are on the order of $1x10^{-10}$ cm/sec (Koerner, 1994), and estimates of the hydraulic conductivity of Gundseal® GCLs are on the order of $1x10^{-12}$ cm/sec (Eith et al., 1991). Essentially all infiltration that does occur through such a composite barrier is the result of defects from manufacturing and / or construction processes. Theoretical performance of the cover was evaluated using the Hydrologic Performance of Landfill Performance (HELP) computer model (Schroeder et al., 1994). HELP is a quasi-two-dimensional, deterministic, water-routing model for determining water balances (Schroeder et al., 1994). HELP predicted 0.000000 inches of annual percolation through the barrier layer. Clearly, this prediction is unrealistic as no cover is absolutely impermeable. Because the performance of the cover system is so closely linked to construction QA/QC, it is very difficult to make an accurate estimate of anticipated infiltration through the barrier layer. It is accurate to state, however, that if this proposed cover system is constructed with appropriate QA/QC, it will meet and exceed the regulatory performance specifications. To accurately monitor the performance of the cover system, it is recommended that the downgradient groundwater quality be closely monitored before and after cover construction to reveal contaminant concentration trends indicative of cover system effectiveness.

While the primary objective of the cover system is to minimize infiltration into the waste fill, there are several other significant performance criteria which must be satisfied. Given the site-specific conditions, the cover system must also:

* isolate the waste from humans, vectors and other animals, and other components of the surrounding ecosystem
* control gases generated within the waste fill
* be resistant to erosion by wind and water
* be resistant to static and seismic slope failures
* be durable, maintaining its design performance level for 30 years (regulatory) or the life of the waste fill (prudent)
* control surface water runoff and lateral drainage flow in a manner which does not promote erosion and does not adversely impact the surrounding environment

As presented in Appendix A-6, these criteria are satisfied by the proposed cover design. The waste is well isolated from the surrounding ecosystem by a total of over 5 feet of soil. Any gases produced by the waste will be vented to atmosphere to prevent explosive conditions from occurring within the waste layer. Additionally, atmospheric monitoring is included as part of the post-closure program to ensure that vented gases do not violate Clean Air Act standards and to ensure that no gas migrates off-site. The cover is designed to be erosion-resistant. The surface is graded to a moderate slope, seeded with an appropriate grass mixture, and covered with straw mulch. Surface water runoff and lateral drainage flow are handled by a network of open channels and culverts which divert flow to specified recharge areas in a controlled manner which also assists in erosion control. The cover system is also resistant to static and seismic slope failure. The minimum static factor of safety of the proposed cover system is 3.1, the minimum seismic factor of safety is 1.0. The recommended minimum factors of safety are 1.5 and 1.0 respectively. It should be noted that it is relatively rare to have a cover design satisfy the seismic stability safety factor in a seismically active area such as Cape Cod. The issue of durability is not so clearly satisfied, in the author's opinion. Relatively little research on the long-term durability of geosynthetics in landfill covers has been performed, and since the history of geosynthetics in cover systems is fairly short, there are few, if any, case studies of sufficient length (e.g., over 30 years) to fill the data gap. However, the research that has been performed indicates that a cover system is an environment which is relatively conducive to geosynthetic survivability (Koerner et al., 1991). In a cover, the geosynthetics are not exposed to toxic chemicals, they are isolated from ultraviolet radiation, and they are fairly well protected from the effects of freeze/thaw cycles. Thus, it seems likely that the cover system will maintain its integrity well into the future.

In summary, it is contended that the proposed cover system will adequately contain the source of the LF-1 plume. If constructed with appropriate construction QA/QC, the proposed cover system design will provide a nearly impermeable barrier while also controlling lateral drainage flow, surface runoff, and decomposition gases with a stable, durable design that should maintain its integrity for decades.

4.2 Bioremediation

Bioremediation of the LF-1 plume has been considered as a potential remedial action for the site, but a comprehensive plan has yet to be proposed (ABB Environmental, 1992). Conventional enhanced bioremediation systems stimulate microbial degradation by amending groundwater from the aquifer with oxygen and nutrients and recirculating it through the contaminated area (O'Brien & Gere Engineers Inc., 1995). The immense size of the LF-1 plume would necessitate the pumping and recirculation of hundreds of millions of gallons of water in order to ensure the removal of all of the chlorinated solvents. This plan would not only be prohibitively costly, it would also be ineffective because the plume contains PCE which cannot be aerobically degraded (Pavlostathis and Zhuang, 1993).

In order to solve the technical problems associated with a traditional enhanced bioremediation action, a passive anaerobic/aerobic system can be used. This system would consist of two groups of horizontal injection wells which are driven into the aquifer at a depth just below that of the plume (see Figure 4-2). The wells would be driven across the width of the plume and have thousands of small injection ports along the top of each one. The ports are used to inject gases into the aquifer in order to stimulate the microbes which will degrade the plume contaminants. Each set of wells will form a distinct biozone above it. The first biozone will be anaerobic and will treat the PCE in the plume, while the second biozone will be an aerobic treatment phase which will remove the remaining chlorinated solvents. This system has a significant advantage over traditional systems because it is a flow-through system; the gas is injected below the plume where it can rise up into the contaminated water and stimulate microbial activity as the plume flows over the gas injection wells. This significantly reduces the pumping costs associated with a more traditional bioremediation system.

The LF-1 plume contains significant quantities of PCE which can only be degraded anaerobically because methanotrophic bacteria possess a monooxygenase enzyme which cannot oxidize a fully chlorinated ethene molecule (Semprini, 1995). Therefore, the first stage of the system must be designed to turn the system anaerobic so that anaerobic bacteria can utilize the PCE in the plume in the process of reductive dechlorination. PCE is an oxidized chemical species while organic matter is relatively reduced. Reductive dechlorinating bacteria use the PCE as a chemical oxidant in a redox reaction with organic matter in order to obtain energy to function and grow (Hollinger et al, 1993). In the process, one or more chlorines are removed from the PCE and replaced with hydrogen. This renders the PCE susceptible to aerobic attack.

In order to turn the aquifer anaerobic, methane and air are injected at the first biozone. This injection serves a threefold purpose. Methanotrophs utilize the methane for growth and deplete the oxygen in the plume as it flows past the well. In addition, the methanotrophs will also degrade some of the TCE and DCE in the plume since their monooxygenase enzymes can degrade the solvents as well as methane (Semprini, 1995). Finally, as methane is utilized by the methanotrophs for growth, biomass will be accumulated in the region above the treatment well. This biomass will then be used by methanogenic bacteria to fuel the process of reductive dechlorination of PCE within the plume.

Once the oxygen is depleted from the plume, the first biozone will be anaerobic. It will remain anaerobic since there will be little or no vertical mixing with oxygenated recharge water (Domenico and Schwartz, 1990). Furthermore, oxygen will be depleted from the plume as it flows into the biozone by periodic injections of methane. Bacteria in this anaerobic zone will utilize the dead biomass and reductively dechlorinate the solvents in the plume. This is a slow biological process; based on laboratory batch studies and the temperature and pH of the aquifer the biozone needs to produce at least five milligrams per liter of biomass and it should take about 540 days to achieve extensive removal (greater than 99 percent) of the PCE in the plume (see appendix A7). Given a PCE migration rate within the plume of .9 ft per day and a treatment zone of two hundred feet associated with each horizontal well, three six-thousand foot horizontal wells will need to be installed to create the first biozone. Some of the TCE and DCE in the plume will also be dechlorinated within this area, rendering all of the chlorinated solvents in the LF-1 plume more susceptible to treatment by aerobic degradation.

The second biozone will be an aerobic zone that will be used to degrade the bulk of the chlorinated solvents in the plume. Gaseous methane, air, nitrous oxide, and triethyl phosphate will be injected into the aquifer (Skiadas, 1996). Methanotrophs will feed on this and will also degrade the solvents in a process termed cometabolic oxidation. One horizontal well must be used to produce the aerobic biozone which will achieve a ninety-five percent reduction in the concentration of TCE and ensure total remediation of DCE and VC (See Appendix A7). This level of remediation is more than sufficient to ensure that federal MCLs for the pollutants in the LF-1 plume are not exceeded in private drinking wells in the path of the plume.

It is apparent that the enhanced bioremediation system proposed above has the potential to effectively remediate the chlorinated solvent plume emanating from the main base landfill at the MMR on Cape Cod. The system would be difficult to manage and expensive to emplace, but it does offer many cost advantages over other remediation or containment schemes because it does not involve pumping large volumes of water or treating contaminated groundwater with granular activated carbon to remove the chlorinated organics. However, this type of system has never been used in the field so a pilot-scale study should be conducted at a smaller site to ensure that the concept works and is cost-effective. If this test produces positive results, then a sequential anaerobic/aerobic enhanced bioremediation system of this nature could be used to clean up the LF-1 plume.

Figure 4-2. Conceptual Diagram of Proposed Bioremediation Scheme (Skiadas, 1996)

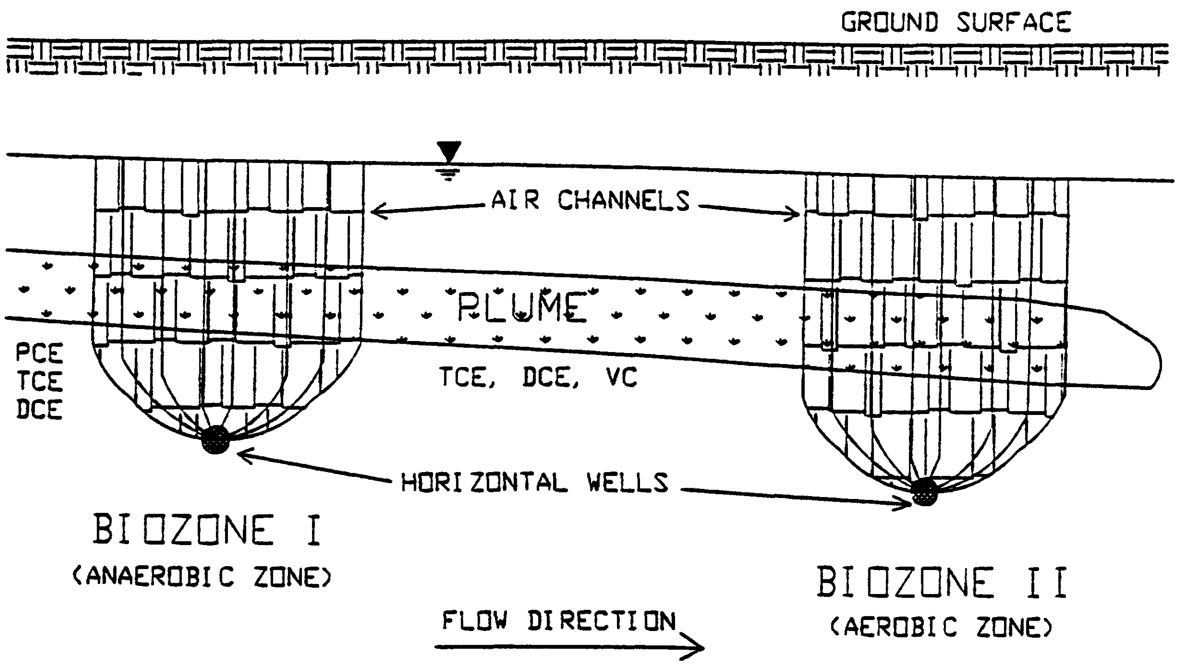


FIGURE 5: CONCEPTUAL DIAGRAM OF THE SEQUENTIAL BIOREMEDIATION SYSTEM (SKIADAS, 1996)

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Appendix A. Supporting Study Reports

1. Site Characterization

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2. Groundwater Modelling

Amarasekera, K.N., "A Groundwater Model Of The Landfill Site At The Massachusetts Military Reservation", M.Eng. Thesis, Massachusetts Institute of Technology, Cambridge, Massachusetts, 1996.

3. Risk Assessment

Hines, J.J., "Uncertainty of Risk to Human Health from Groundwater Impacted by the Massachusetts Military Reservation Superfund Site Landfill", M.Eng. Thesis, Massachusetts Institute of Technology, Cambridge, Massachusetts, 1996.

4. Risk Assessment

Lee, R.F., "Human Health and Coastal Ecosystem Risk Assessment of the Massachusetts Military Reservation Main Base Landfill Groundwater Plume", M.Eng. Thesis, Massachusetts Institute of Technology, Cambridge, Massachusetts, 1996.

5. Management

Jordan, B.R., "Management and Public Interaction Surrounding the Massachusetts Military Reservation Installation Restoration Program", M.Eng. Thesis, Massachusetts Institute of Technology, Cambridge, Massachusetts, 1996.

6. Source Containment

Elias, K.G., "Source Containment at the Massachusetts Military Reservation Main Base Landfill: Design of a Hazardous Waste Landfill Cover System," M.Eng. Thesis, Massachusetts Institute of Technology, Cambridge, Massachusetts, 1996.

7. Bioremediation

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