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Groundwater Remediation Strategies at the Streator Former Manufactured Gas Plant Site

A Case Study Using the Remedial Options Assessment Model (ROAM)

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PRODUCT DESCRIPTION

This report describes the application of EPRI's Remedial Options Assessment Model (ROAM™ Version 1.0) at a former manufactured gas plant (MGP) site in Streator, Illinois. The model was used to assess current trends in groundwater contaminant concentrations at the site and to evaluate remedial alternatives.

Results and Findings

This report presents a case study whose elements are applicable to MGP sites slated for regulatory closure. If the Streator site owners implement no further action beyond pump-and-treat, or rely on natural attenuation, ROAM results show that contaminant concentrations in the groundwater immediately north of the nearby Vermillion River could remain higher than State of Illinois Surface Water Quality Standards (SWQS) for more than 30 years. Aggressive transformation of the residual contamination will be necessary to lower groundwater concentrations to meet SWQS in a relatively short timeframe of 5 to 10 years. Aggressive transformation involves 10 times the natural degradation rate (10 x natural attenuation) for benzene and 50 times the natural degradation rate (50 x natural attenuation) for naphthalene.

Challenges and Objectives

Since many electric utility companies are responsible for properly closing former MGP facilities, company owners and personnel who manage these sites will benefit from information provided in this report.

Site closure presents a challenge at the Streator former MGP site. In 2001, owners installed a pump-and-treat system that successfully intercepts contaminated groundwater, preventing discharge to the nearby Vermilion River. However, contaminant concentrations in water collected from the recovery trench do not meet State of Illinois SWQS, as required by the Illinois Environmental Protection Agency. It is clear that site closure will not be achieved without further remediation. This report shows how using ROAM to model the site demonstrates that regulatory closure will be possible and provides a framework for further investigation. Narrowing remedial options with ROAM can help the companies avoid costs associated with implementing ineffective solutions.

Applications, Values, and Use

As electric utility companies develop former MGP sites or offer them for sale, they must assure that the sites meet regulatory requirements governing acceptable concentrations of contaminants in soil, groundwater, and surface water. Transport and fate modeling that predicts the impact of remedial actions meets this need. EPRI continues to support and upgrade ROAM for this purpose.

EPRI Perspective

EPRI's ROAM is specially designed to simulate soil and groundwater transport of contaminants from chemical mixtures such as MGP tars and to predict the effect of remedial actions on contaminant plume migration. This focus makes ROAM uniquely valuable as a tool to help owners make sound, cost-effective decisions about remediation at MGP sites.

Approach

The goal of this research was to identify a remedial option that would achieve regulatory closure at the Streator former MGP site within a timeframe of 5 to 10 years. Using site-specific data and literature values for physical and chemical parameters, the project team calibrated ROAM to site conditions until model results indicated a reasonable match with measured benzene and naphthalene concentrations in groundwater and water in the recovery trench. The team then used ROAM to model alternatives including no action, natural attenuation, 10 x natural attenuation, and 50 x natural attenuation.

Keywords

Manufactured Gas Plant
ROAM™
Benzene
Naphthalene
Site Closure

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1

SITE BACKGROUND

The former manufactured gas plant (MGP) site is a 4.2-acre lot located along the north bank of the Vermilion River in Streator, Illinois. Gas was produced from coal at the plant between 1876 and 1933. According to records in *Brown's Directory of American Gas Companies* [1], coal carbonization was the primary process used at the plant from 1876 until 1914, followed by the Lowe water-gas process from 1914 to 1933. After 1933, the plant no longer produced manufactured gas and was used only as a natural gas storage facility.

Site Investigation and Remediation

A thorough site characterization, including evaluation of the extent of MGP impacts to soil and groundwater, was performed between 1987 and 1999. Results of the investigations indicated that MGP source material was present in underground foundations of former plant structures, and soil and groundwater were impacted with MGP residuals. Also, tar seeps in the form of nonaqueous-phase liquid (NAPL) and dissolved constituents in groundwater were observed along the riverbank. The site layout, including former plant structures, is shown in Figure 1-1.

In 1998, prior to implementing a final remedy for the site, the owners removed 1,541 tons of nonhazardous source material and 731 tons of hazardous source material from two gas holders, a pump house, a tar well, and a vertical tank. The final remedy includes an engineered barrier placed over impacted soils to prevent direct exposure to those soils, and a groundwater containment system to prevent impacted groundwater and NAPL from discharging to the Vermilion River. Constructed in 2000, the engineered barrier consists of 3 feet of clean soil cover spread over 2.3 acres. The containment system constructed along the northern bank of the Vermilion River between 2000 and 2001 consists of a 500-foot long, 3-foot wide and 20-foot deep recovery trench and barrier wall with three sumps (Figure 1-2). Groundwater from the site flowing toward the Vermilion River is captured in the trench and treated by carbon adsorption before being released to the publicly owned treatment works. An average pumping rate of 10 gallons per minute (gpm) is maintained to achieve hydraulic control of the contaminated groundwater.

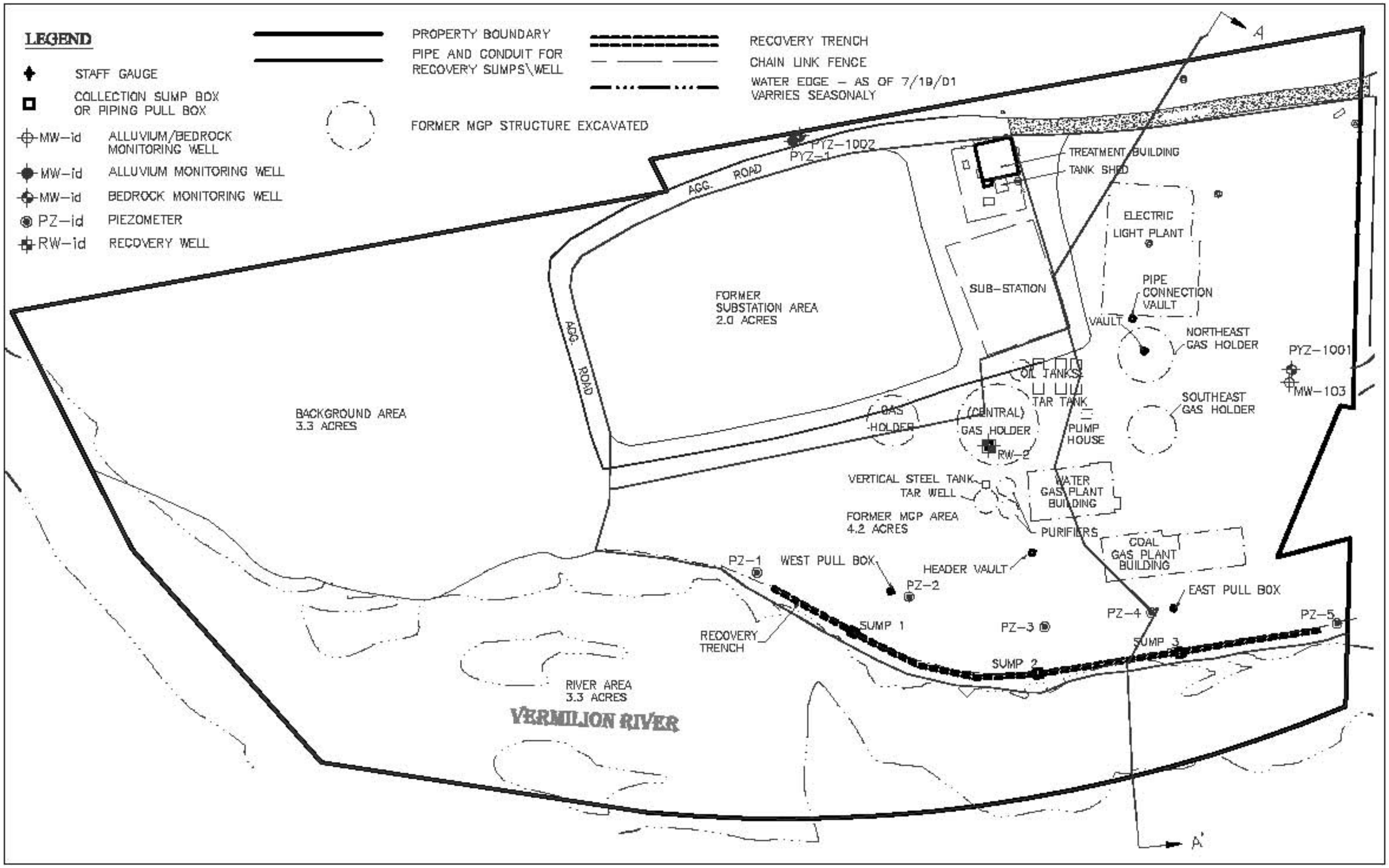

Figure 1-1
Current and Historic Site Layout

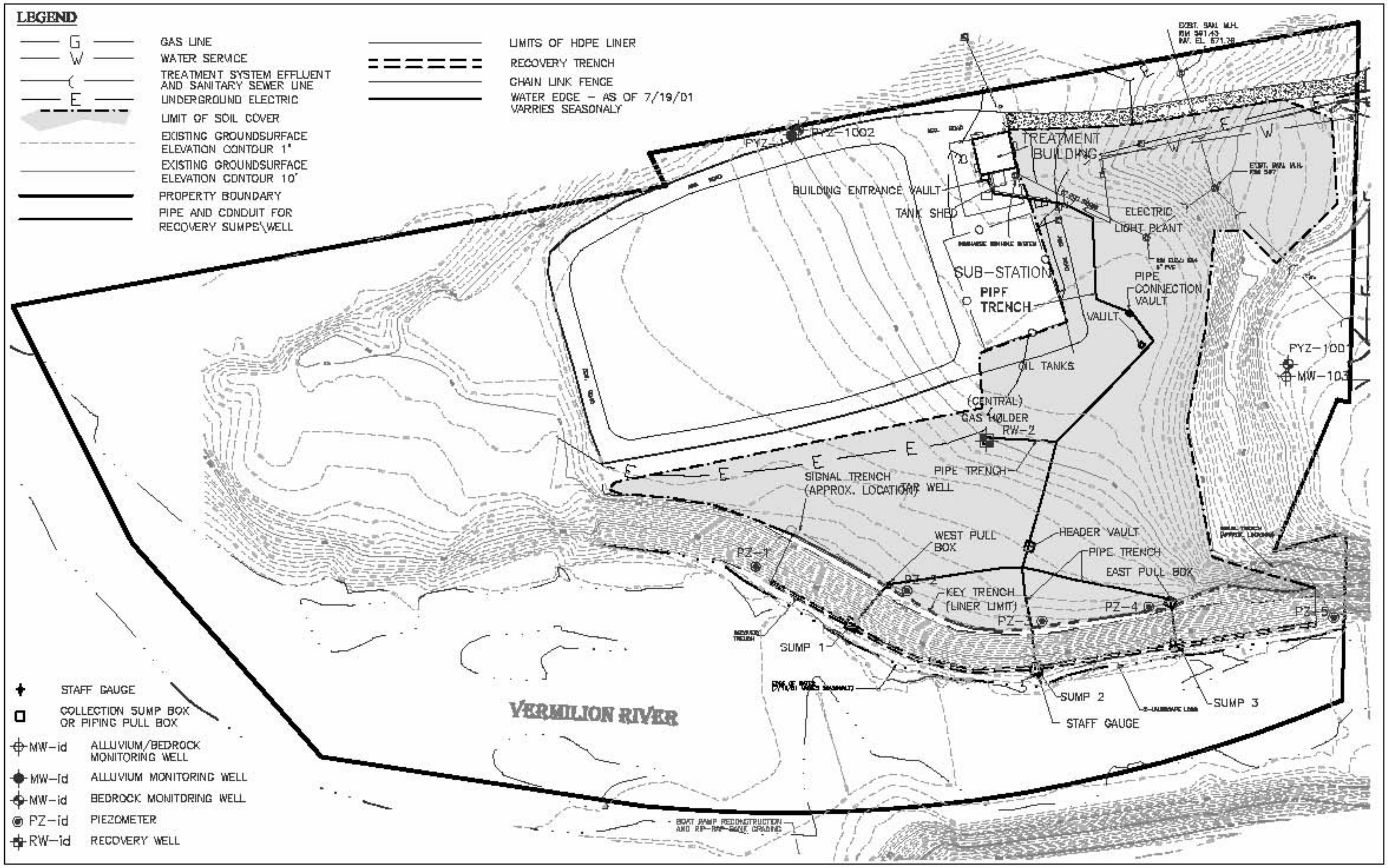

Figure 1-2
Groundwater Containment System Layout

Nature and Extent of Contamination

In 1998, source material samples were collected from the two gas holders, the pump house, and the tar well. Benzene and naphthalene were used as indicator chemicals to assess the level of site contamination. Benzene concentrations ranged from non-detect at the pump house to 1,100 mg/kg at the northeast gas holder. Naphthalene concentrations ranged from 27 mg/kg at the pump house to 13,000 mg/kg at the northeast gas holder.

Samples were also collected from nine monitoring wells in the bedrock aquifer at or near the source. Average benzene and naphthalene concentrations in these samples were 5.63 and 13.83 mg/l (5,630 and 13,830 µg/l), respectively. Influent concentrations of benzene and naphthalene in the trench have been recorded on a monthly basis since September 2001. Contaminant concentrations in the trench have fluctuated, as can be seen in Figure 1-3. The average benzene and naphthalene concentrations between 2001 and 2007 were 1.06 and 0.50 mg/l (1,060 and 500 µg/l), respectively.

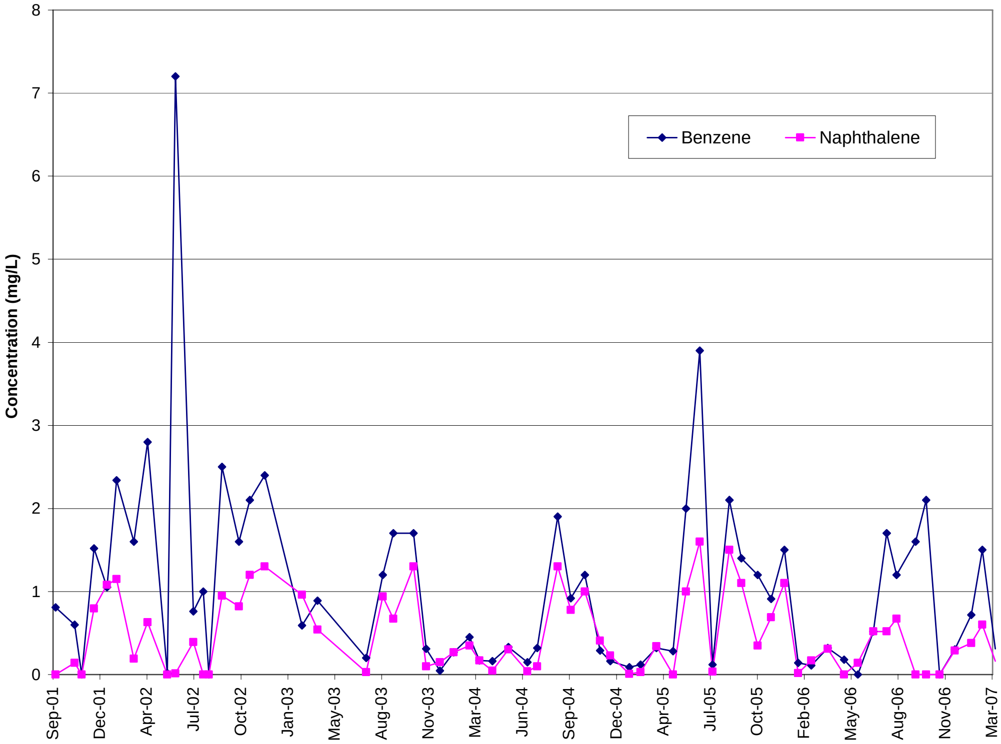

Figure 1-3
Benzene and Naphthalene Concentrations in Trench, September 2001 to March 2007

It is important to note that concentrations in the trench may not represent the actual groundwater concentrations in the aquifer, which are expected to be higher than concentrations in the trench.

Geology

The site generally exhibits two types of unconsolidated soil deposits: upland deposits 40 feet thick in the eastern portion of the site, and alluvial deposits ranging from 5 to 18 feet thick in the central and western portions of the site. The unconsolidated depositional units are underlain by sandstone bedrock approximately 12 to 18 feet thick. The sandstone is fine-grained and often weathered and/or fractured in its upper few feet. The sandstone matrix becomes finer with depth as it grades into shale. The underlying shale is often weathered and/or fractured in its upper few feet. A north–south (A–A') geological cross section of the site is shown in Figure 1-4. The location of the geologic cross section is shown in Figure 1-1.

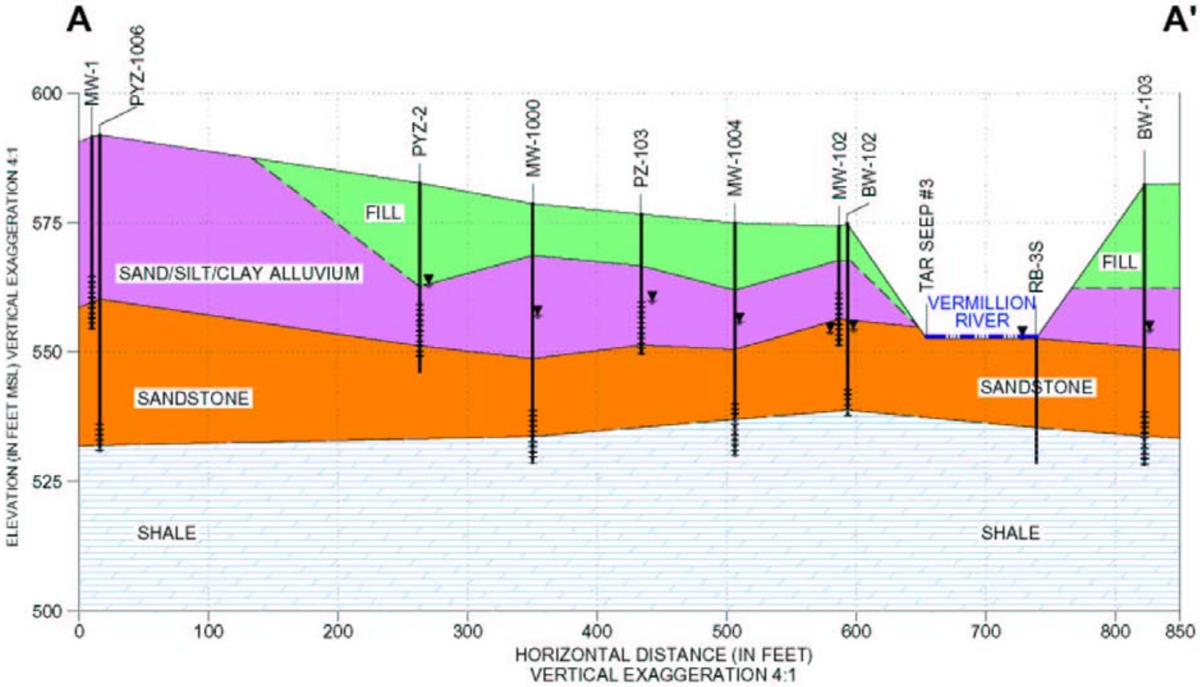

Figure 1-4
North–South Geological Cross Section with Vertical Exaggeration

South of the site, the sandstone forms the bed of the Vermilion River. In the river, the sandstone ranges in thickness from approximately 13 feet at locations parallel to the western portion of the site (downstream) to approximately 20 feet at locations east of the site (upstream).

Hydrogeology

Two shallow aquifers underlie the site. The upper aquifer (alluvium/bedrock aquifer) includes the lower alluvium and the upper portion of the sandstone bedrock. The lower aquifer (bedrock aquifer) includes the middle and lower portions of the sandstone bedrock, and the upper 1 to 2 feet of highly fractured underlying shale. Based on slug tests and pump tests, the hydraulic conductivity geometric means for the alluvium/bedrock formation and the bedrock formation are estimated to be 1.22×10^{-2} cm/s (34.5 ft/d) and 6.77×10^{-4} cm/s (1.92 ft/d), respectively. Depth to the water table ranges from 12 to 23 feet below ground surface. The general direction of flow for both the alluvium/bedrock and bedrock aquifers is south to southwest, toward the Vermilion River.

Objective of ROAM Study

From 2001 to the present, a pump-and-treat system has operated at the Streator former MGP site. It has successfully intercepted contaminated groundwater to prevent discharge to the Vermilion River. However, contaminant concentrations in water collected from the recovery trench do not meet State of Illinois Surface Water Quality Standards (SWQS), as required by the Illinois Environmental Protection Agency.

Thus, the objective of this project was to assess current trends in contaminant concentrations at the site, and to evaluate alternative groundwater remediation strategies using the Remedial Options Assessment Model (ROAM™ Version 1.0) developed by EPRI. The goal was to identify a remedial option that would achieve regulatory site closure within a timeframe of 5 to 10 years.

2
MODEL APPLICATION

ROAM Description

ROAM is a multi-component flow and transport model that predicts soil and groundwater contaminant concentrations before and after remediation. It evaluates complex fate and transport processes in both the saturated and unsaturated (vadose) zones. Multiple sources of contamination can be simulated at the same time in either zone.

In the vadose zone, ROAM uses either a single contaminant analytical solution or a multi-contaminant numerical solution to perform fate and transport modeling. In the saturated zone, ROAM uses either a three-dimensional analytical solution or a two-dimensional numerical solution. For the latter, ROAM combines a two-dimensional numerical transport model with a two-dimensional numerical steady-state flow model to predict contaminant flow and transport. Multiple sources of contamination in the saturated and/or vadose zones—and their potential impacts on groundwater quality—are coupled to produce an overall solution.

ROAM's Applicability to MGP Sites

ROAM was specially designed and developed to aid decision making during MGP site characterization and implementation of remedial options. The model can simulate the fate and transport of most MGP-related contaminants. In light contamination, multiple chemicals are present in moderate amounts, but the fate and transport of the chemical of interest is not affected by their presence. In heavy contamination, multiple chemicals—such as fuels and/or coal tar/oil by-products—are present, and the fate and transport of the chemical of interest is affected by their presence. To deal with the complexity created by variation in the number of chemical constituents and their proportions at an MGP site, ROAM incorporates a built-in database with an extensive collection of fate- and transport-related data on MGP contaminants.

Model Setup

Site Conceptual Model

In the site conceptual model shown in Figure 2-1, the alluvium/bedrock formation and the bedrock formation act as a single hydrogeologic unit under unconfined (water table) flow conditions. The coal oil/tar initially present in unsaturated soil seeps downward into the groundwater and migrates laterally. The general direction of groundwater flow is south to southwest, toward the Vermilion River.

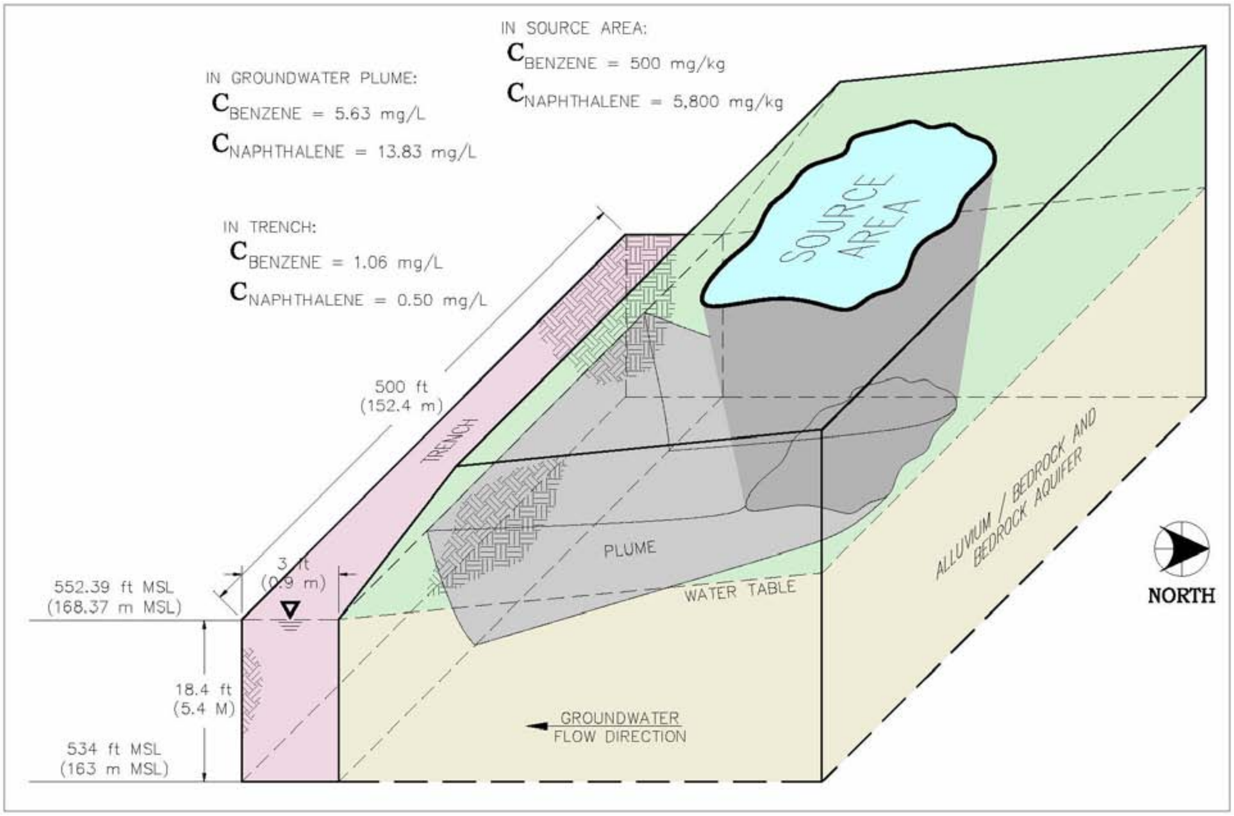

Figure 2-1
Site Conceptual Model

Model Construction

A grid of squares, each measuring 4 m x 4 m (16 m^2), covering a total area of 240 m x 240 m (57,600 m^2), was generated to simulate the former MGP site and to focus on the main areas of concern. The source area was estimated to be 60 m x 60 m (197 ft x 197 ft). A thickness of 20 feet (6.1 m) was used to represent the combined saturated thickness of the alluvium/bedrock and bedrock formations. It was assumed that the source initially occupied the vadose zone, which was 22.5 ft (6.9 m) thick in the estimated source area. The modeling grid system superimposed on the site layout map is shown in Figure 2-2.

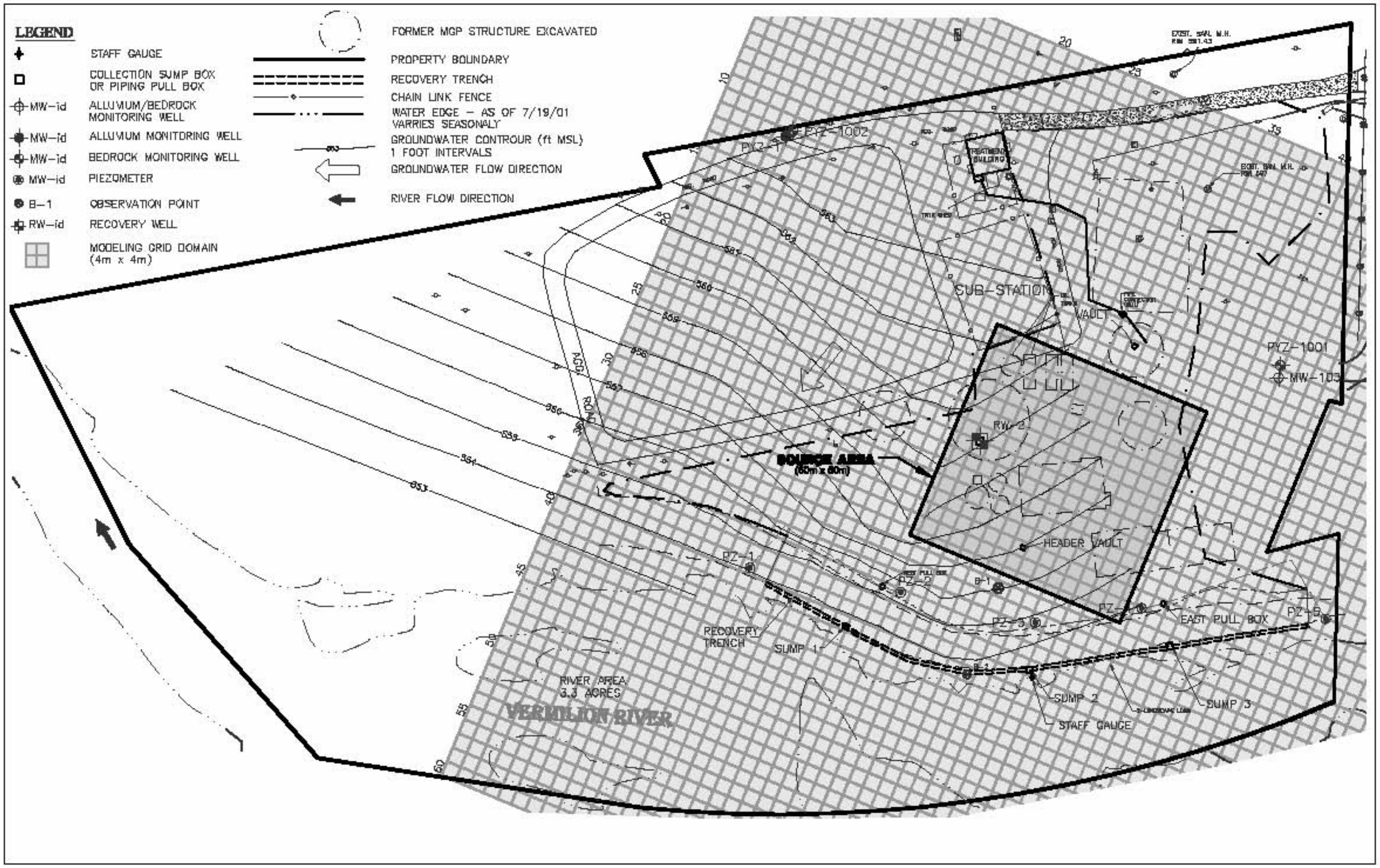

Figure 2-2
Modeling Grid System

A simple north to south flow of groundwater was assumed for the model. To simulate the currently operating recovery trench, Drains 1 and 2 were added to the site in a V-like shape downgradient of the former MGP process area. Piezometer PZ-3 was added to aid in generating the flow field and to enable groundwater contaminant concentration monitoring upgradient of the trench. Piezometer PZ-3 was located south of the source area and upgradient of the recovery trench, as shown in Figure 2-3. To ensure that model boundary conditions remained unaffected by site features such as source area and drains, a buffer zone of 50 to 100 m was added between the main site features and the outer limits of the model.

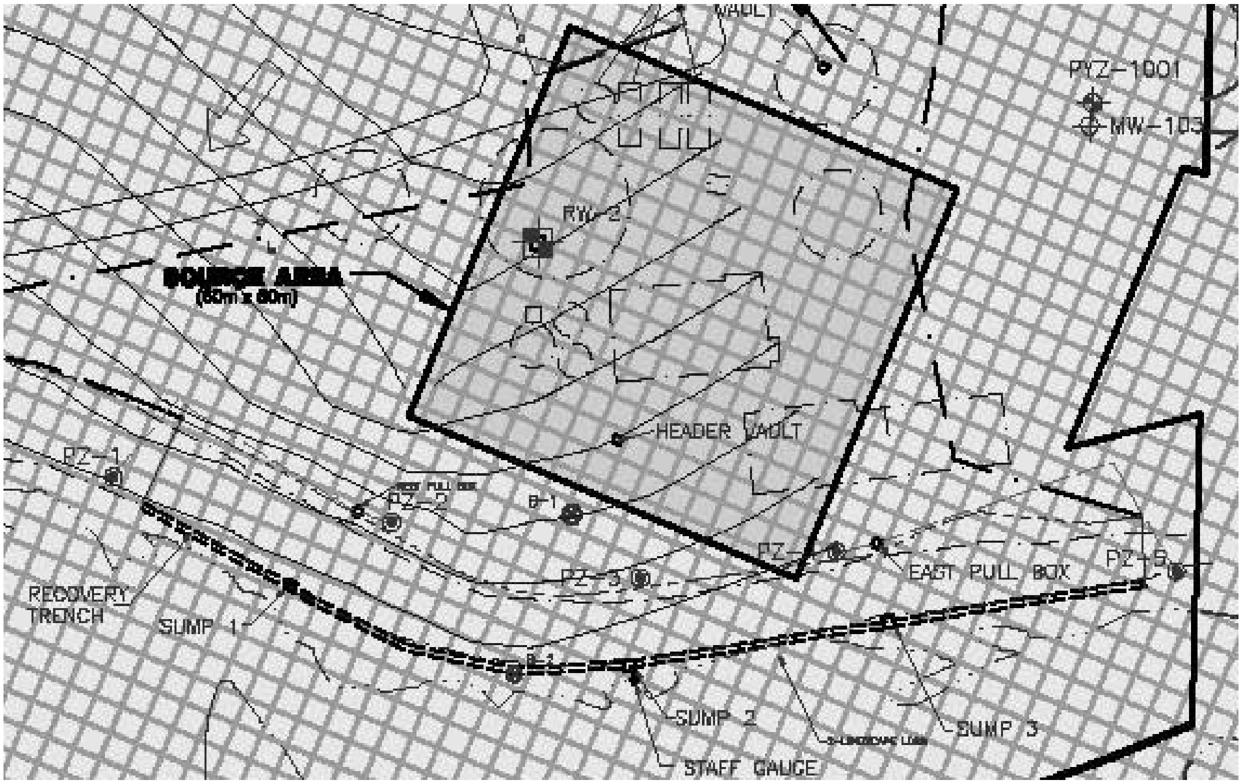

Figure 2-3
Detail of Modeled Area

Boundary Conditions

Boundary conditions in the modeled area were defined by constant head boundaries along the boundaries perpendicular to the flow direction and no flow boundaries along the boundaries parallel to the flow direction. The constant head at boundaries along the northern and southern boundaries of the modeled area was calculated by extrapolating the specified groundwater elevation at piezometer PZ-3, taking into account an aquifer recharge of 0.1 m/yr (3.94 in/yr). Details on calculating boundary heads are presented in the ROAM user's manual [2].

Input Parameters

Site-specific geological and hydrogeological parameters were used as model inputs. Table 2-1 lists parameters used for model calibration and/or flow and transport modeling. Initially, the hydraulic conductivity for the bedrock aquifer was used to calibrate the flow model (Section 3). Seepage velocity was calculated from the hydraulic conductivity, the effective porosity, and the observed hydraulic gradient. A reference groundwater elevation of 173 m MSL (568 ft MSL) at piezometer PZ-3 was used to generate the groundwater flow field.

Table 2-1
Geological and Hydrogeological Model Input Parameters

Parameter	Values	Units
Alluvium/Bedrock Aquifer Hydraulic Conductivity	1.22×10^{-2} (34.5)	cm/s (ft/d)
Bedrock Aquifer Hydraulic Conductivity	6.77×10^{-4} (1.92)	cm/s (ft/d)
Trench Flow Rate	57.34 (9.86)	m^3/d (gpm)
Water Level in Trench	168.37 (552.39)	m MSL (ft MSL)
Total Porosity	0.35	–
Effective Porosity	0.25	–
Hydraulic Gradient	0.017	m/m
Net Precipitation Recharge Rate	0.1 (3.94)	m/yr (in/yr)
Dry Bulk Density	1.65	g/cm^3
Organic Carbon Content	1.35	%

ROAM Site Evaluation

Current site data were used to predict the migration potential of coal tar, based on educated assumptions and a scientific approach using ROAM. Site evaluation using ROAM predicted contaminant behavior under a no action alternative and various remedial alternatives.

The No Action Alternative

In the no action alternative, the pump-and-treat system continues to operate as usual and no additional remedial actions are taken. The no action alternative is executed after construction of the model and input of all necessary parameters. Output from the no action alternative provides information about the timeframe required to meet regulatory standards without remedial intervention, assuming that natural degradation does not play an important role at the site. This information can be used to determine the necessity of implementing a remedial alternative to hasten contaminant reduction and achieve site closure.

Remedial Alternatives

ROAM can be used to evaluate various remedial alternatives for multiple sources of contamination in the vadose or saturated zones. Vadose zone alternatives might include excavation, capping, or tank removal. Saturated zone alternatives might include biotransformation, or installation of groundwater extraction wells, drains, or slurry walls.

Soil and groundwater quality data can be projected into the future via contour plots and installed monitoring points to assess the viability of each remedial alternative.

3

GROUNDWATER FLOW MODEL CALIBRATION

Approach and Trial Runs

Groundwater flow model calibration is the process of adjusting selected model parameters within an expected range until differences between model-predicted quantities and field-observed quantities match selected criteria for performance. Calibration is generally conducted against the observed heads of the aquifer and/or flow rates to hydraulic structures such as drains. At the Streator former MGP site, the main groundwater flow is toward the Vermilion River. The trench runs parallel to and immediately north of the river. The flow rate to the trench is a direct function of the aquifer hydraulic conductivity and net precipitation recharge to the aquifer. Therefore, calibration with the trench flow rate data is sufficient for modeling purposes.

Groundwater flow model calibration for the Streator site was performed using the hydraulic data listed in Table 2-1. The average measured flow rate to the trench of 9.86 gpm (53.74 m^3/d) was compared to calculated flow rates to the trench. ROAM provides groundwater contours as outputs, not numerical flow rates. Thus, flow rate was calculated from the head contours map, with some approximations. Horizontal hydraulic conductivity (K_h) of the aquifer was the parameter exerting the most influence on flow rate to the trench. For the calibration process, the goal was to adjust the K_h value until flow rate to the trench predicted by ROAM was approximately equal to the observed flow rate to the trench (9.86 gpm). Several trial runs were conducted to accomplish this.

Trial Run 1

For this trial run, the geometric mean value of the hydraulic conductivity $K_h = 6.77 \times 10^{-4}$ cm/s (214 m/yr) was used as the representative value of the aquifer. The effective porosity value, $n_e = 0.25$, and the average hydraulic gradient, $i = 0.017$ m/m, were used. The calculated seepage velocity was 14.55 m/yr. The net precipitation recharge for this run was $R = 0.1$ m/yr. Using the model steady state head distribution results adjacent to the trench to calculate the hydraulic gradient and applying Darcy's law, the flow rate to the trench was $Q_{trench} = 25.15\ m^3/d$ (4.61 gpm). This value was smaller than the observed flow rate of 53.74 m^3/d (9.86 gpm). Due to the difference between the observed and predicted flow rates to the trench, additional adjustment was required. Calculation details are given in Appendix A.

Trial Run 2

For this trial run, the hydraulic conductivity value was increased to $K_h = 1.35 \times 10^{-3}$ cm/s (427 m/yr). The effective porosity value, $n_e = 0.25$, and the average hydraulic gradient,

$i = 0.017$ m/m, remained the same as before. The calculated seepage velocity was 29.04 m/yr. The net precipitation recharge for this run, $R = 0.1$ m/yr, remained the same. Using the model steady state head distribution results adjacent to the trench and applying Darcy's law, the flow rate to the trench was $Q_{trench} = 62.42$ m^3/d (11.45 gpm). This value was reasonably close to the observed flow rate of 53.74 m^3/d (9.86 gpm). Calculation details are given in Appendix A.

4

CONTAMINANT TRANPORT MODEL CALIBRATION AND REMEDIAL ALTERNATIVES

Input Data for the Contaminant Transport Model

In selecting input data for contaminant transport modeling, tar source material at the site was assumed to date from 1905, the median year of operations between 1876 and 1933. The source was assumed to fully occupy the vadose zone, which was estimated to be 6.9 m (22.5 ft) thick. The calculated seepage velocity of the calibrated flow model, V = 29.04 m/yr, was used in the contaminant transport model. Organic carbon content and dry bulk density included in Table 2-1 were also part of contaminant transport modeling. The longitudinal dispersivity (α_L), transverse dispersivity (α_T), and vertical dispersivity (α_V) were estimated based on the scale-dependent approach presented in Gelhar, et al. [3]. Assuming that the distance between the center of the source and the trench is 60 m, the longitudinal dispersivity was estimated from Figure 2 in Gelhar et al. (longitudinal dispersivity versus scale with data classified by reliability) as $\alpha_L = 4$ m. The transverse dispersivity and the vertical dispersivity are calculated as $\alpha_T = 0.1\alpha_L = 0.4$ m and $\alpha_V = 0.01\alpha_L = 0.04$ m, respectively.

Contaminant Model Calibration Approach

ROAM requires specification of source characteristics in the vadose zone. The required source characteristics are: type of source, contaminant constituents, and constituent concentrations in mg/kg units. The type of source at the Streator site was "tar" and the selected constituents were benzene, naphthalene, and total petroleum hydrocarbon (TPH). Due to limited source concentration and distribution data from the site, measured groundwater concentrations in the bedrock aquifer were used to determine the source concentrations and dimensions. Benzene and naphthalene were chosen as representative MGP-related contaminants in the groundwater.

The source concentrations of benzene and naphthalene in the vadose zone, along with the horizontal dimensions of the source area, were changed until the simulated groundwater concentrations reasonably matched the measured average groundwater concentrations. The average benzene and naphthalene concentrations in the bedrock aquifer in 1998 were 5.63 and 13.83 mg/l (5,630 and 13,830 ug/l), respectively. The average benzene and naphthalene concentrations in the trench between 2001 and 2007 were 1.06 and 0.50 mg/l (1,060 and 500 ug/l), respectively. Due to mixing in the trench water body, these values were lower than the corresponding groundwater concentrations at the discharge point to the trench.

Based on average concentrations in four former on-site MGP structures, the initial benzene and naphthalene concentrations in the source area were estimated to be 500 and 5,800 mg/kg, respectively. The TPH source concentration for each of its components (gasoline, diesel, and heavy hydrocarbons) was chosen to be 500 mg/kg. The initial estimated source dimensions were 60 m (197 ft) wide, 60 m (197 ft) long, and 6.9 m (22.5 ft) deep, based on the location of the former MGP structures and the average thickness of the vadose zone. For all computer runs, the average thickness of the source remained 6.9 m (22.5 ft), but horizontal dimensions and the source concentrations were varied.

ROAM was used to model a no action alternative and three remedial alternatives summarized in Table 4-1. The no action alternative excludes remedial intervention and assumes that natural degradation does not play an important role at the site. The remedial alternatives include natural attenuation, 10 x natural attenuation, and 50 x natural attenuation.

The model requires that the dissolved biotransformation rate and adsorbed biotransformation rate be specified. In the no action alternative, these biotransformation rates were set to zero. Since the dissolved phase is the major form subject to degradation, the adsorbed biotransformation rate was set to zero for all remediation alternatives. In natural attenuation, the dissolved biotransformation rate was the natural degradation rate. In the other two remedial alternatives—which enhance natural attenuation—the dissolved biotransformation rates were 10 and 50 times the natural degradation rate, respectively. Under natural attenuation, biotransformation was assumed to begin in 1905, when the tar source was introduced into the vadose zone. Under enhanced natural attenuation, biotransformation was assumed to begin in 2007.

Table 4-1
Summary of Modeled Alternatives

Modeled Alternatives		Dissolved Biotransformation Rate	Adsorbed Biotransformation Rate	Biotransformation Starts
No Action Alternative		0	0	___
Remedial Alternatives	Natural Attenuation	Natural Degradation Rate	0	1905
	10 x Natural Attenuation	10 x Natural Degradation Rate	0	2007
	50 x Natural Attenuation	50 x Natural Degradation Rate	0	2007

For each model run, calculated benzene and naphthalene concentrations in the groundwater were plotted versus time at observation points B-1 and B-2. Observation point B-1 is located at the centerline of the plume along the southern boundary of the source area, at approximately 34 m (115.5 ft) from the center of the source area. Observation point B-2 is located along the centerline of the plume adjacent to the trench, at approximately 58 m (190 ft) from the center of the source area. The locations of observation points B-1 and B-2 are shown in Figure 2-3.

Based on the trial runs, it was determined that reasonable agreement between calculated and measured groundwater concentrations was given by a source width of 40 m (131 ft) and length of 60 m (197 ft), a benzene source concentration of 300 mg/kg, and a naphthalene source concentration of 10,000 mg/kg. Averaged benzene and naphthalene concentrations for the year 1998 (as measured in monitoring wells in and around the source area) and for the years 2001 to 2007 (as measured in the trench) were compared with calculated concentrations in groundwater. In the following sections, benzene and naphthalene are evaluated separately for each modeled alternative.

Benzene Concentrations

The average degradation rate for benzene under natural conditions is 0.0009 d^{-1} (0.3285 yr^{-1}) [4]. The corresponding half-life is 770 days. The minimum SWQS for benzene is 310 µg/l [5].

No Action

A plot of depth-averaged benzene concentration versus time at observation points B-1 and B-2 is shown in Figure 4-1 for the no action alternative. The calculated benzene concentration increases sharply from 1905, reaching maximum values of 5,715 µg/l at observation point B-1 in 1932 and 5,414 µg/l at observation point B-2 in 1937. The calculated benzene concentration then declines slowly, reaching 4,567 and 4,450 µg/l at observation points B-1 and B-2, respectively, in 2040.

For the year 1998, the calculated benzene concentration at observation point B-1 is 5,365 µg/l. This value is similar to the averaged benzene concentration of 5,630 µg/l measured in monitoring wells in and around the source area. For the years 2001 to 2007, the calculated benzene concentration at observation point B-2 is 5,020 µg/l. This value is higher than the averaged benzene concentration of 1,060 µg/l measured in the trench. This discrepancy can be attributed to mixing and dilution that may occur in the trench water body.

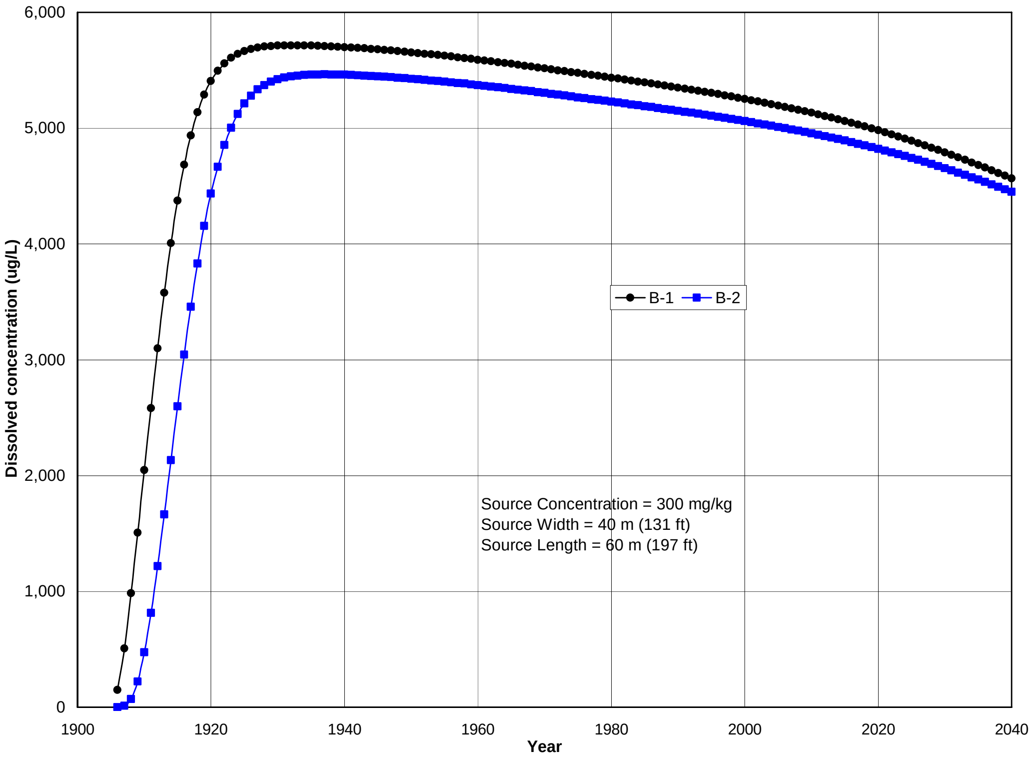

Figure 4-1
Depth-Averaged Benzene Concentration versus Time at Observation Points B-1 and B-2 under No Action

Natural Attenuation

In this alternative, the dissolved biotransformation rate is 0.3285 yr^{-1}, which is equal to the natural degradation rate. Assuming that natural degradation has occurred since 1905 when the tar source was introduced into the vadose zone, Figure 4-2 shows that the calculated benzene concentration increases sharply to maximum values of 3,724 µg/l at observation point B-1 in 1930, and 2,776 µg/l at observation point B-2 in 1934. After that, the calculated benzene concentration declines slowly, reaching 2,956 µg/l and 2,243 µg/l at observation points B-1 and B-2, respectively, in 2040. Under natural attenuation, the benzene concentration at observation point B-2—located adjacent to the trench at approximately 58 m (190 ft) from the center of the source area—is still significantly above the SWQS (310 µg/l) in 2040.

The benzene profiles under no action and natural attenuation are very similar (see Figures 4-1 and 4-2). However, benzene concentrations under no action are higher than those under natural attenuation. This difference increases with the distance from the source area but decreases with time. As a result, for the year 1998 the calculated benzene concentration at observation point B-1 is 3,487µg/l. This value is somewhat lower than the averaged benzene concentration of 5,630 µg/l measured in monitoring wells in and around the source area. For the years 2001 to 2007, the calculated benzene concentration at observation point B-2 is 2,541µg/l. This value is somewhat higher than the averaged benzene concentration of 1,060 µg/l measured in the trench.

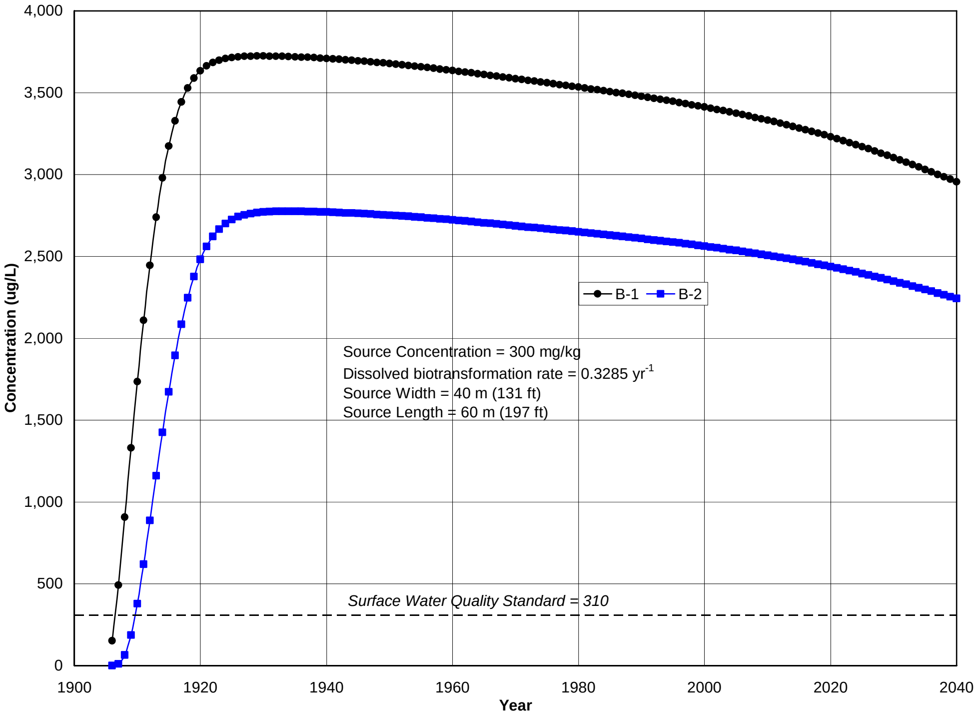

Figure 4-2
Depth-Averaged Benzene Concentration versus Time at Observation Points B-1 and B-2 under Natural Attenuation

10 x Natural Attenuation

In this alternative, the dissolved biotransformation rate is 3.285 yr^{-1}, which is 10 times the natural degradation rate (0.3285 yr^{-1}). As shown in Figure 4-3, after biotransformation begins in 2007, the calculated benzene concentration at observation points B-1 and B-2 drops sharply from 5,171µg/l and 4,989 µg/l to approximately 600 µg/l and 300 µg/l, respectively, in 2011. It appears that about 4 years after enhanced biotransformation begins, the benzene concentration at observation point B-2—located adjacent to the trench at approximately 58 m (190 ft) from the center of the source area—drops below the SWQS (310 µg/l).

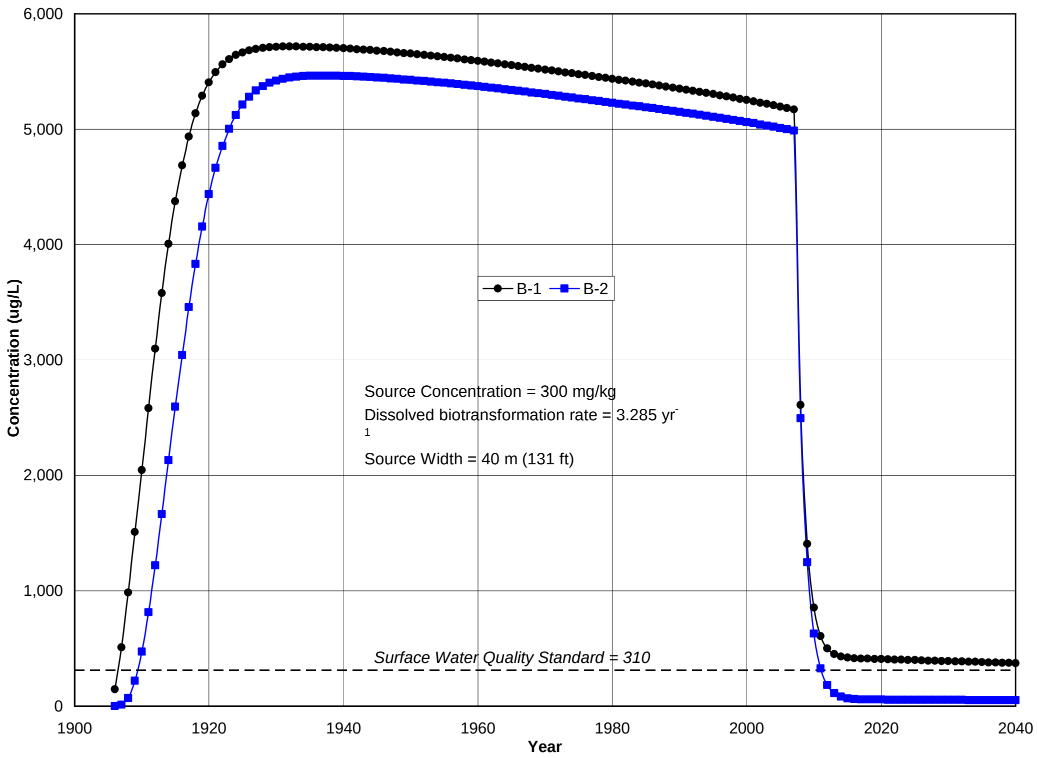

Figure 4-3
Depth-Averaged Benzene Concentration versus Time at Observation Points B-1 and B-2 under 10 x Natural Attenuation

50 x Natural Attenuation

In this alternative, the dissolved biotransformation rate is 16.425 yr^{-1}, which is 50 times the natural degradation rate (0.3285 yr^{-1}). As shown in Figure 4-4, after biotransformation begins in 2007, the calculated benzene concentration at observation points B-1 and B-2 drops sharply from 5,171 µg/l and 4,989 µg/l to approximately 200 µg/l and 150 µg/l, respectively, in 2008. It appears that about 1 year after enhanced biotransformation begins, the benzene concentration at observation point B-2—located adjacent to the trench at approximately 58 m (190 ft) from the center of the source area—drops below the SWQS (310 µg/l).

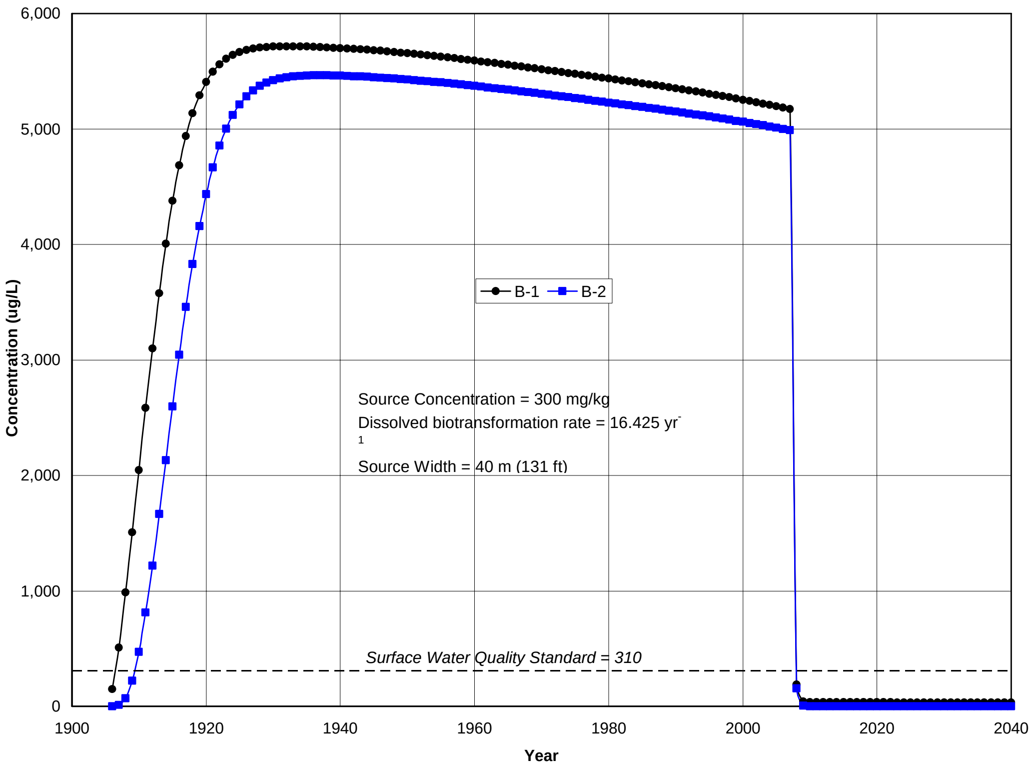

Figure 4-4
Depth-Averaged Benzene Concentration versus Time at Observation Points B-1 and B-2 under 50 x Natural Attenuation

Naphthalene Concentrations

The average degradation rate for naphthalene under natural conditions is 0.0027 d^{-1} (0.9855 yr^{-1}) [4]. The corresponding half-life is 256.7 days. The minimum SWQS for benzene is 68 μg/l [5].

No Action

A plot of depth-averaged naphthalene concentration versus time at observation points B-1 and B-2 is shown in Figure 4-5 for the no action alternative. The calculated naphthalene concentration increases gradually from 1905 to 1935, reaching approximately 370 μg/l at observation point B-1. During this period, the calculated naphthalene concentration remains near zero at observation point B-2. The calculated naphthalene concentration continues to increase with time, reaching 3,132 μg/l and 786 μg/l at observation points B-1 and B-2, respectively, in 2040.

For the year 1998, the calculated naphthalene concentration at observation point B-1 is 1,678 μg/l. This value is significantly smaller than the averaged naphthalene concentration of 13,830 μg/l measured in monitoring wells in and around the source area. However, due to the likely presence of free product in the wells, 3 of 11 measured concentrations were above 10,000 μg/l. If these data are excluded, the average measured naphthalene concentration is

3,510 µg/l, which is similar to the calculated value of 1,678 µg/l. For the years 2001 to 2007, the calculated naphthalene concentration at observation point B-2 is 286 µg/l. This value is quite similar to the averaged benzene concentration of 500 µg/l measured in the trench.

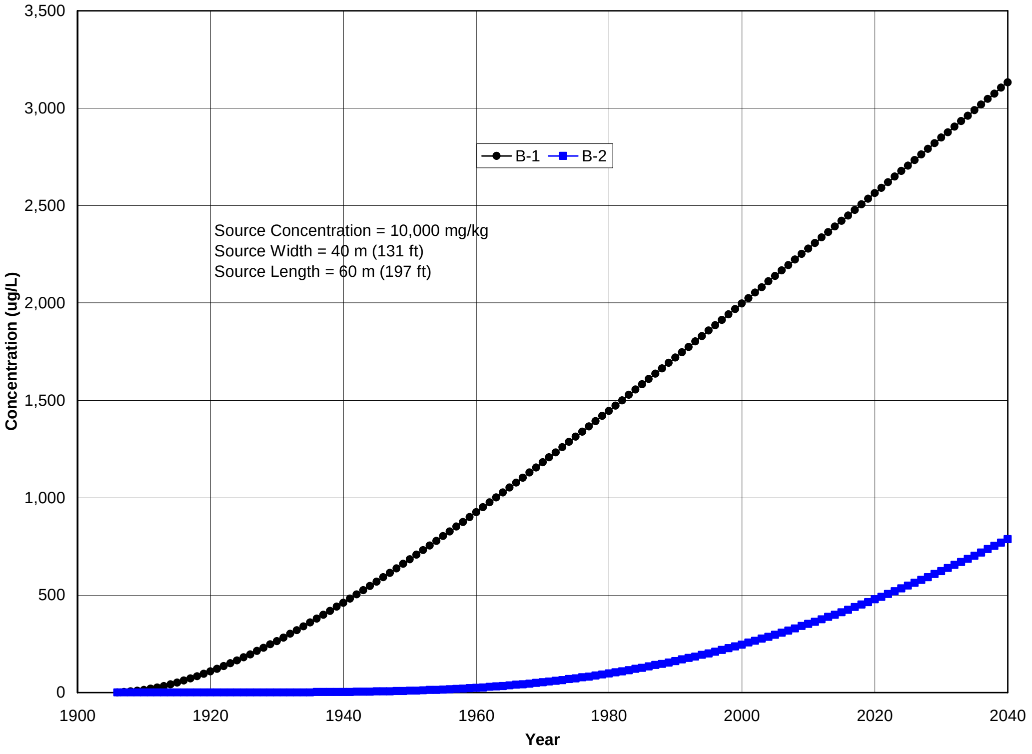

Figure 4-5
Depth-Averaged Naphthalene Concentration versus Time at Observation Points B-1 and B-2 under No Action

Natural Attenuation

In this alternative, the dissolved biotransformation rate is 0.9855 yr^{-1}, which is equal to the natural degradation rate. Assuming that natural degradation has occurred since 1905 when the tar source was introduced into the vadose zone, Figure 4-6 shows that the calculated naphthalene concentration increases gradually in the next 30 years to approximately 350 µg/l at observation point B-1, but remains near zero at observation point B-2. After that, the calculated naphthalene concentration continues to increase with time, reaching 1,834 µg/l and 372 µg/l at observation points B-1 and B-2, respectively, in 2040. Under natural attenuation, the naphthalene concentration at observation point B-2—located adjacent to the trench at approximately 58 m (190 feet) from the center of the source area—is still significantly above the SWQS (68 µg/l) in 2040.

Naphthalene profiles under no action and natural attenuation are very similar (see Figures 4-5 and 4-6). However, naphthalene concentrations under no action are higher than those under natural attenuation. This difference decreases with distance form the source area but increases with time. As a result, for the year 1998 the calculated naphthalene concentration at observation point B-1 is 1,196 μg/l. This value is significantly lower the averaged naphthalene concentration of 13,830 μg/l measured in monitoring wells in and around the source area. However, as discussed above, the measured naphthalene concentration in the monitoring wells may be elevated due to the presence of free product. A more realistic naphthalene concentration of 3,510 μg/l is somewhat closer to the calculated one. For the years 2001 to 2007, the calculated naphthalene concentration at observation point B-2 is 164 μg/l. This value is somewhat lower the averaged naphthalene concentration of 500 μg/l measured in the trench.

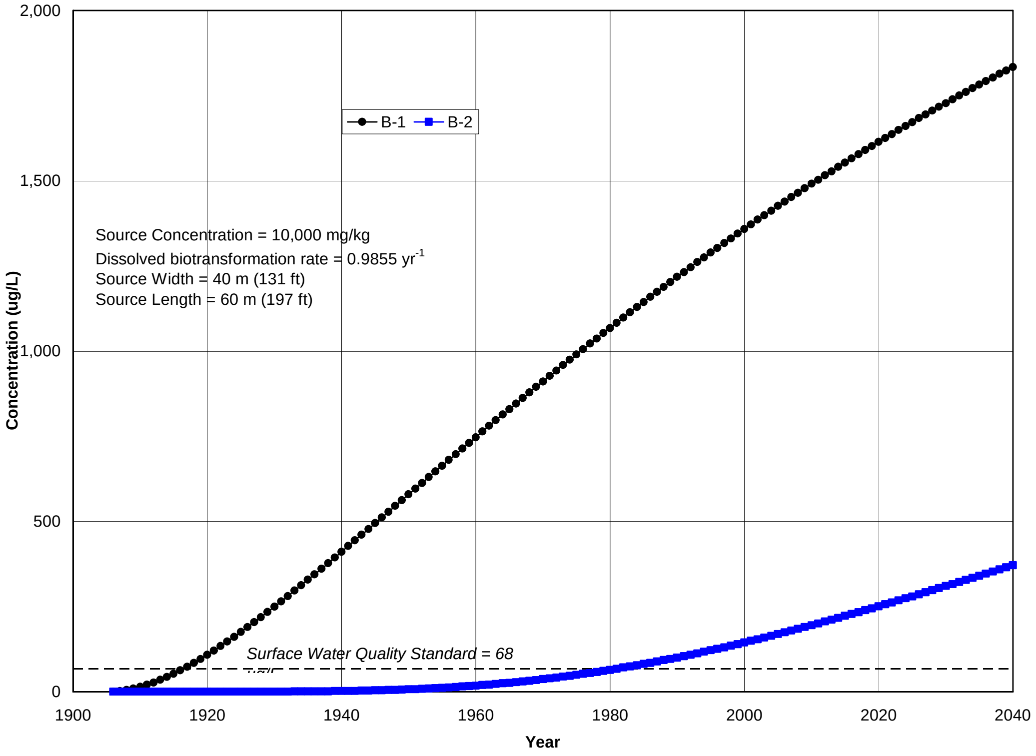

Figure 4-6
Depth-Averaged Naphthalene Concentration versus Time at Observation Points B-1 and B-2 under Natural Attenuation

10 x Natural Attenuation

In this alternative, the dissolved biotransformation rate is 9.855 yr^{-1}, which is 10 times the natural degradation rate (0.9855 yr^{-1}). As shown in Figure 4-7, after biotransformation begins in 2007, the calculated naphthalene concentration at observation points B-1 and B-2 drops rapidly from 2,194 μg/l and 318 μg/l to approximately 300 μg/l and 60 μg/l, respectively, in 2040. It appears that about 33 years after enhanced biotransformation starts, the naphthalene concentration at observation point B-2—located adjacent to the trench at approximately 58 m (190 feet) from the center of the source area—drops below the SWQS (68 μg/l).

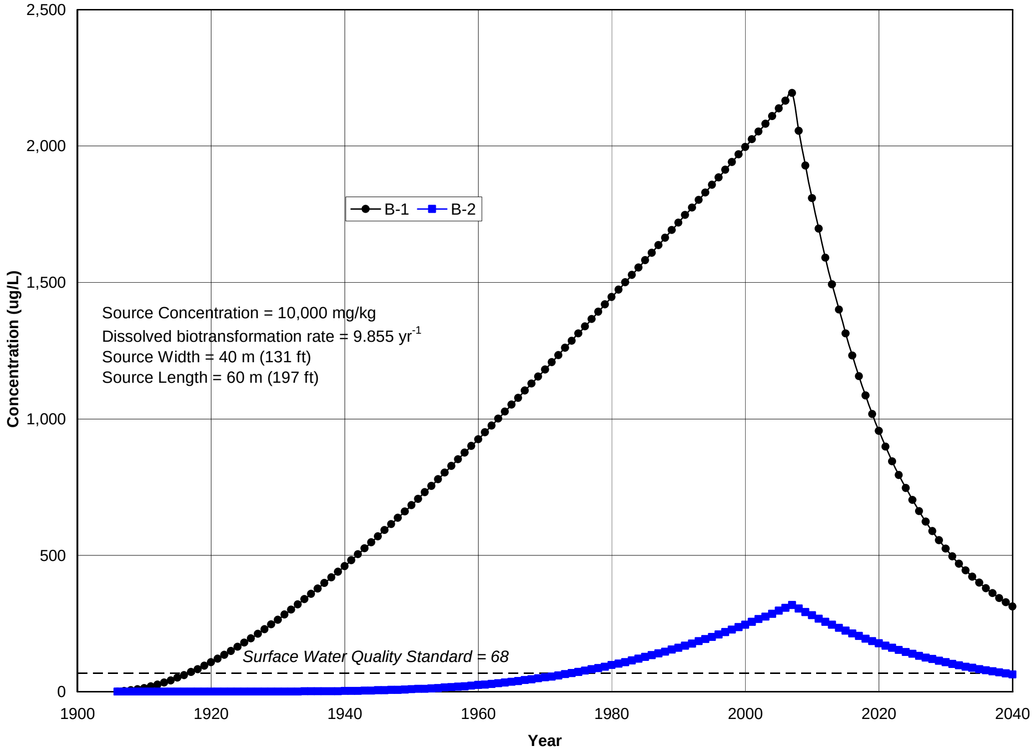

Figure 4-7
Depth-Averaged Naphthalene Concentration versus Time at Observation Points B-1 and B-2 under 10 x Natural Attenuation

50 x Natural Attenuation

In this alternative, the dissolved biotransformation rate is $49.275 yr^{-1}$, which is 50 times the natural degradation rate (0.9855 yr^{-1}). As shown in Figure 4-8, after biotransformation begins in 2007, the calculated naphthalene concentration at observation points B-1 and B-2 drops sharply from 2,194 µg/l and 318 µg/l to approximately 350 µg/l and 55 µg/l, respectively, in 2012. It appears that approximately 5 years after enhanced biotransformation begins, the naphthalene concentration at observation point B-2—located adjacent to the trench at approximately 58 m (190 feet) from the center of the source area—drops below the SWQS (68 µg/l).

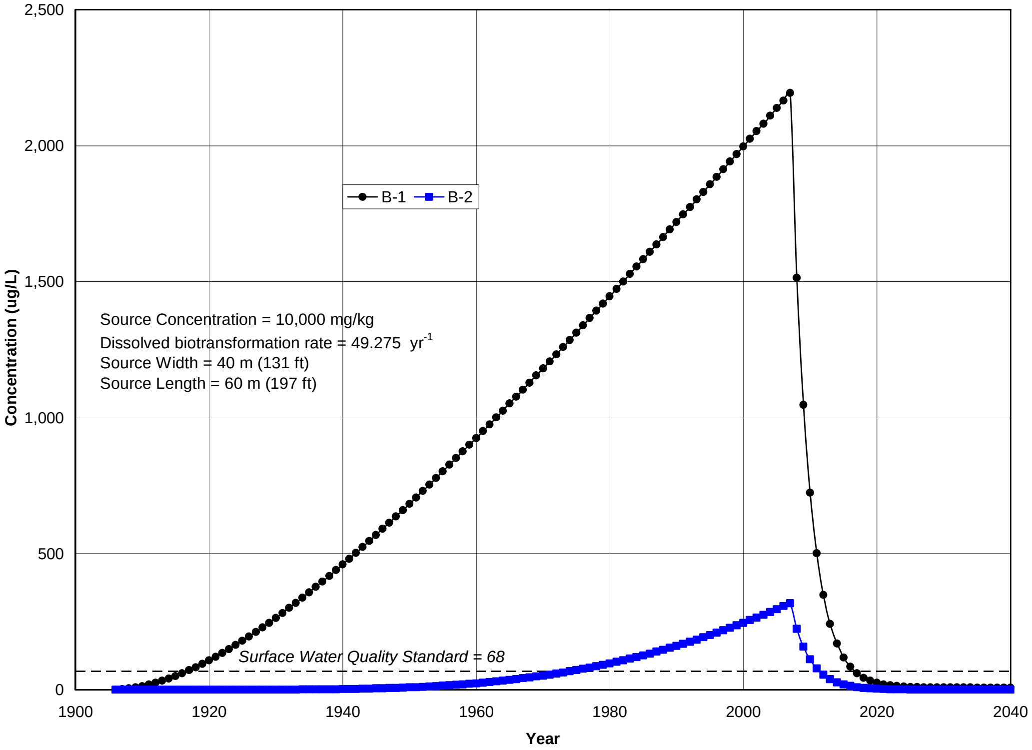

Figure 4-8
Depth-Averaged Naphthalene Concentration versus Time at Observation Points B-1 and B-2 under 50 x Natural Attenuation

5

DISCUSSION

ROAM was successfully used to assess current trends in groundwater contaminant concentrations at the Streator MGP site and to evaluate remedial alternatives. The model was constructed on a grid of squares, each measuring 4 m x 4 m (16 m^2), covering a total area of 240 m x 240 m (57,600 m^2). The initial estimate of the source area—60 m (197 ft) wide x 60 m (197 ft) long—was based on the location of the former MGP structures. It was assumed that the source occupied the vadose zone, estimated to be 22.5 ft (6.9 m) thick under the source area. The initial estimates of benzene (500 mg/kg) and naphthalene (5,800 mg/kg) concentrations in the source area were based on the measured average benzene and naphthalene concentrations in four former MGP structures.

To specify site conditions that reasonably replicated measured groundwater concentrations, initially estimated source area dimensions were revised during model calibration. The width of the source area was adjusted from 60 to 40 m, but the length of the source area remained 60 m and its thickness remained 6.9 m. Similarly, as part of the calibration process, the benzene source concentration was lowered from an estimated 500 to 300 mg/kg, and the naphthalene source concentration was elevated from an estimated 5,800 to 10,000 mg/kg. Under these conditions, model results indicated a reasonable match with measured groundwater concentrations and water concentrations in the trench.

Benzene and naphthalene behaved differently, as shown in the depth-averaged concentration versus time curves. Under no action or natural attenuation, benzene concentrations in the groundwater increased rapidly, reached a peak, and then slowly decreased. On the other hand, naphthalene concentrations in the groundwater increased gradually and did not peak. Naphthalene has a higher biotransformation rate than benzene, which means that naphthalene is more likely to be transformed once it is in solution. However, naphthalene is nearly two orders of magnitude less soluble than benzene, which retards its dissolution into groundwater.

It is clear from both actual site data and model results that natural attenuation alone will not reduce the groundwater contamination at this site to State of Illinois SWQS levels in less than 30 years. Aggressive transformation of the residual contamination would be necessary to lower groundwater contaminant levels to meet SWQS in a relatively short timeframe. Aggressive transformation involves 10 times the natural degradation rate (10 x natural attenuation) for benzene and 50 times the natural degradation rate (50 x natural attenuation) for naphthalene.

Although ROAM relies heavily on a biodegradation constant to demonstrate contaminant reduction, this constant can be viewed broadly as contaminant transformation. In addition to biodegradation, contaminant transformation can mean destruction or immobilization of contaminants, implying that technologies other than bioremediation can be used to obtain the modeled results.

6
CONCLUSIONS AND RECOMMENDATIONS

Conclusions

Based on ROAM results, the following conclusions can be drawn:

- Under present pump-and-treat conditions (no action) or natural attenuation, contamination in the groundwater at observation point B-2—located adjacent to the trench immediately north of the Vermilion River—will remain at concentrations above State of Illinois SWQS for over 30 years.
- If contaminants were to be degraded or otherwise destroyed at a rate 10 times the current natural degradation rate (10 x natural attenuation), benzene and naphthalene concentrations at observation point B-2 would meet SWQS in 4 and 33 years, respectively.
- This would not be adequate for site closure within a timeframe of 5 to 10 years.
- If the contaminants were to be degraded or otherwise destroyed at a rate 50 times the current natural degradation rate (50 x natural attenuation), the benzene and naphthalene concentrations at observation point B-2 would potentially meet SWQS in 1 and 5 years, respectively.
- This would result in acceptable water quality discharge to the Vermilion River, and potential site closure, within 5 years.

Recommendations

The following recommendations are proposed as next steps:

- Research alternative remedial technologies with the potential to degrade or destroy MGP contaminants relatively rapidly. Examples might include chemical oxidation, thermal technologies, or their combined use. Evaluate these technologies for possible use at the Streator former MGP site.
- Identify and fill data gaps to provide information that could aid in further defining, and possibly limiting, the size of the area to be remediated.
- Perform bench-scale treatability tests to determine the potential effectiveness of selected technologies and to collect preliminary design information.
- Design pilot or full-scale tests of the most favorable technology, based on treatability test results.

7
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A

APPENDIX—FLOW MODEL CALIBRATION

Trial 1

Effective porosity
$n_e = 0.25$

Total porosity
n = 0.35

Horizontal hydraulic conductivity
K_h = 6.77 x 10-4 cm/sec = 213.5 m/year = 0.585 m/day

Average hydraulic gradient across the site
i = 0.017 m/m

Seepage velocity
V = (213.5 m/year) x (0.017 m/m)/0.25 = 14.55 m/year

Recharge rate
R = 0.1 m/year = 3.94 in/year

Total length of drain
$L_D = L_{D1} + L_{D2}$ = 63.24 m + 90.24 m = 153.48 m = 503.54 ft

Area perpendicular to flow direction
A = (153.48 m) x (168.37m – 163m) = (153.48 m) x (5.4m) = 828.8 m^2

Hydraulic gradient adjacent to drain
i = Δh/ΔL -= (169.2 m -- 168.37 m)/16 m = 0.83 m/16 m = 0.052 m/m

Calculated flow rate to the drain
Q_{drain} = KhiA = (0.585 m/day) x (0.052 m/m) x (828.8 m^2) = 25.15 m^3/day

Q_{drain} = 25.15 m^3/day < $Q_{drain\ meas}$ = 53.74 m^3/day

Trial 2

Effective porosity
ne = 0.25

Total porosity
n = 0.35

Horizontal hydraulic conductivity
K_h = 2 x 6.77 x 10-4 cm/sec = 427 m/year = 1.17 m/day

Average hydraulic gradient across the site
i = 0.017 m/m

Seepage velocity
V = (427 m/year) x (0.017 m/m)/0.25 = 29.04 m/year

Recharge rate
R = 0.1 m/year = 3.94 in/year

Total length of drain
$L_D = L_{D1} + L_{D2}$ = 63.24 m + 90.24 m = 153.48 m = 503.54 ft

Area perpendicular to flow direction
A = (153.48 m) x (168.37m – 163m) = (153.48 m) x (5.4m) = 828.8 m^2

Hydraulic gradient adjacent to drain
i = Δh/ΔL-= (169.40 m - 168.37 m)/16 m = 1.03 m/16 m = 0.064 m/m

Calculated flow rate to the drain
Q_{drain} = KhiA = (1.17 m/day) x (0.064 m/m) x (828.8 m2) = 62.42 m^3/day

Q_{drain} = 62.42 m^3/day > $Q_{drain\ meas}$ = 53.74 m^3/day

Q_{drain} = 11.45 m^3/day > $Q_{drain\ meas}$ = 9.86 m^3/day

This calculated flow rate value is close enough to the measured value.

The seepage velocity value of 29.04 m/year was used in the transport model.

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