5.0 Contamination Transport Modeling

5.0 GROUND WATER CONTAMINANT TRANSPORT MODELING

The objective of the ground water contaminant transport modeling is to develop an analytical tool based on the geologic and geochemical conditions around the PVLF such that current and future concentrations of chemicals in ground water can be reasonably estimated and then used for a baseline health risk assessment. To provide technical support for the risk assessment, hydrogeologic modeling is conducted to simulate the ground water flow, and contaminant transport modeling is conducted to simulate the movement of chemicals between the landfill and potential receptors and to estimate concentrations of chemicals in ground water at those points.

Because of the complex geologic and hydrogeologic conditions at the PVLF, simplifying assumptions are necessary to facilitate mathematical solutions of the transport equations. The assumptions made in the development of the ground water transport model, however, are designed to result in conservative estimates of modeled chemicals such that the model can be used for evaluation of health risks associated with the ground water transport pathway based on the maximum expected concentrations. For example, the ground water monitoring data discussed in Section 3.6 do not indicate leachate formation and release from the PVLF, but the contaminant transport model described in this section assumes continuous release of all modeled water quality parameters and uses this information as an initial condition for long term predictive analysis. The purpose of this unrealistic assumption is to investigate the health risk associated with the worst case scenario of ground water contamination.

It should be recognized throughout this section that the developed contaminant transport model is extremely conservative because of uncertainties associated with model parameters and the complex geology and hydrogeology at the PVLF. It is, however, a useful tool to estimate maximum health risks that can be expected from some chemicals, volatile organic compounds in particular, in the ground water downgradient of the PVLF.

A ground water flow model has been developed by Dames and Moore, Inc., and is described in Section 3.4.5 of this report. The ground water flow model has been calibrated to provide a spatial distribution of ground water velocity for the transport model. The developed ground water contaminant transport model uses the velocity distribution and calculates concentrations for each water quality parameter of interest. The developed contaminant transport model is calibrated by matching the observed temporal and spatial distributions of chemicals with modeled chemical concentrations. Model parameters are adjusted to refine the goodness of fit until the difference between the observed and calculated chemical concentrations is within tolerance limits. Once the model is calibrated, it is validated by comparing the model predicted concentrations with a second set of observed concentrations to ensure the difference is acceptable. The calibrated model is then used as a tool for predictive analysis which serves as the basis for risk assessment. Sensitivity analyses are performed to quantify fluctuations of modeled concentrations caused by uncertainty conditions.

Because information obtained from the ground water contaminant transport model is used in the baseline risk assessment described in Section 6.0, the ground water contaminant transport model focuses on a list of water quality parameters that may pose a potential threat to human health and their transport mechanisms in the ground water. Model development, assumptions, methodology, and key findings from use of the developed model are described in the following sections. Details about the ground water contaminant transport model developed for the remedial investigation study are contained in Appendix E.3.

5.1 TRANSPORT MECHANISMS

Contaminant movement in the ground water is controlled by physical, chemical, and biological processes. The most important processes are:

- Convection or advection;
- Hydrodynamic dispersion;
- Interphase mass transfer (e.g., adsorption/desorption, volatilization);
- Biodegradation; and
- Abiotic chemical transformation.

These processes are discussed in the following paragraphs.

Convection or advection is the dominant process controlling the translational movement of contaminants. It is the component of solute movement attributed to transport by the flowing ground water. The process of hydrodynamic dispersion occurs as a result of mechanical mixing and molecular diffusion, which causes the contaminant plume to disperse and diffuse. Adsorption represents the retardation of movement of the contaminant plume by physico-chemical adherence of contaminants onto the aquifer matrix, including natural organic matter in the aquifer. Desorption involves the transfer of contaminants adsorbed to the aquifer matrix to the fluid phase (i.e., the opposite of adsorption). Volatilization involves the loss of contaminants from the fluid phase to the vapor phase.

Biodegradation involves biological processes whereby contaminants are broken down by microorganisms in ground water to simpler compounds or compounds with smaller molecules. Abiotic chemical transformation can occur through chemical reactions between the contaminant of concern and chemical constituents in the ground water, soil skeleton, and other catalysts which may be present in the soil-water system. This process is generally difficult to measure in the field. Both biodegradation and abiotic chemical transformation tend to attenuate the contaminant plumes (particularly organic constituents).

The transport model constructed in this study includes convection (or advection), hydrodynamic dispersion, adsorption/desorption, and biodegradation. Abiotic chemical transformation is combined with biodegradation because the same mathematical model (first order decay) is used to describe both processes. Volatilization of chemicals is not included in the transport model; this assumption results in conservative estimates of modeled chemical concentrations in the ground water for the purpose of baseline risk assessment.

5.2 MODEL SELECTION

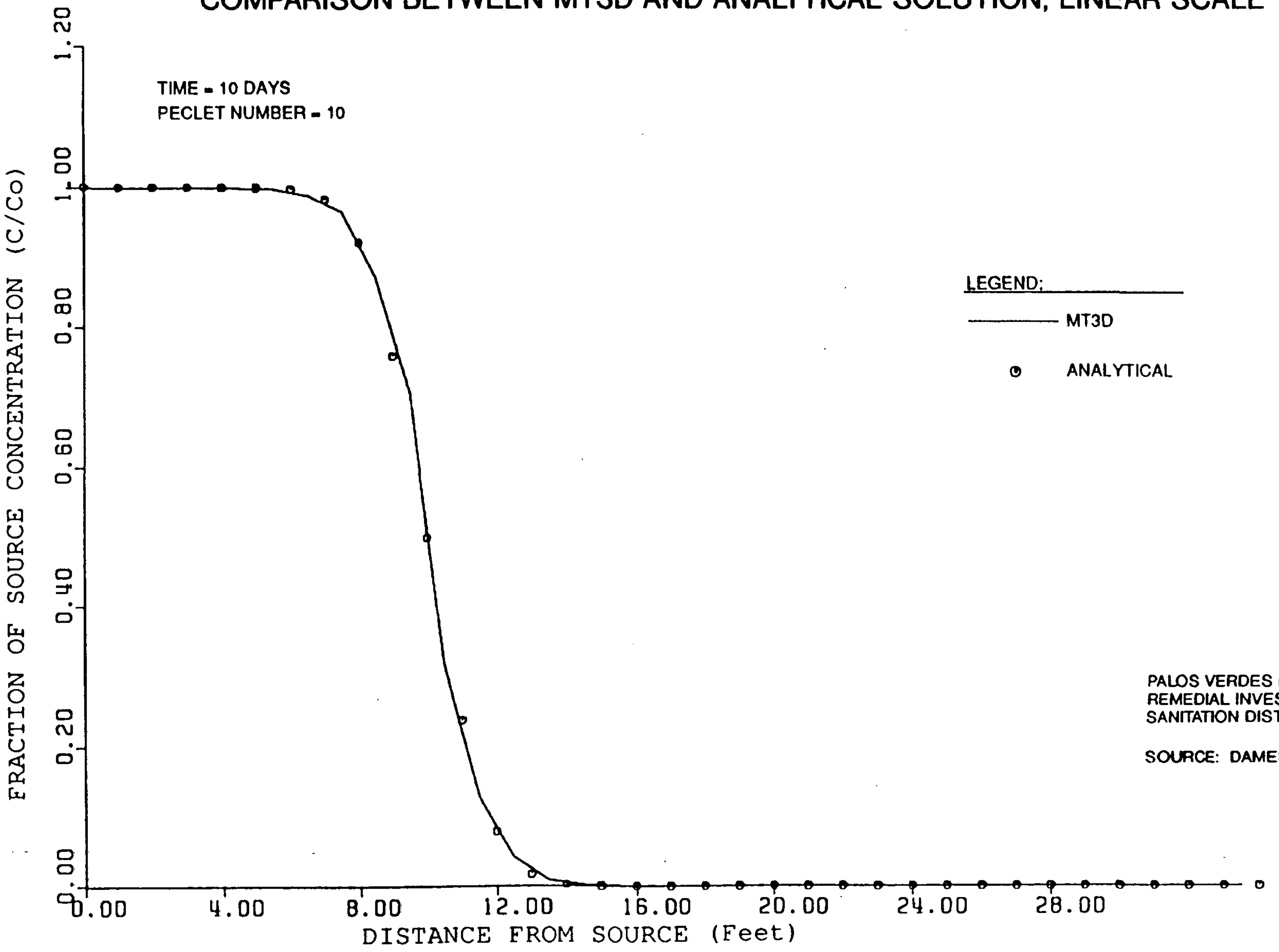
Numerous computer codes are available commercially as well as in the public domain for characterization and simulation of contaminant transport in ground water. The complexity of the hydrogeologic conditions in the vicinity of the PVLF necessitates the use of a three-dimensional contaminant transport model to adequately simulate the behavior of solutes in the subsurface environment.

Following a thorough review and comparison of seven computer codes most frequently used for modeling contaminant transport in ground water, the computer code known as MT3D (Zheng, 1992) was selected for this project. The decision of selecting MT3D was based on: (1) the accuracy of the numerical solution technique used in MT3D; (2) its ability to model free surface (water table) conditions, infiltration at the water table, and irregular-domain configuration; (3) its proven history of success with similar sites for similar purposes; and (4) the model has been verified against analytical solutions and validated with actual field data. The code is fully compatible with the USGS-MODFLOW code (McDonald and Harbaugh, 1988) which was used to simulate the ground water flow. Method of characteristics, which is a numerical solution technique that produces more accurate solutions than do other commonly used numerical methods (such as finite difference and finite element methods) for transport problems with a high Péclet number (ratio of convective flux to dispersive flux; a value of ten is expected at the PVLF) is employed by MT3D. The important transport mechanisms described in Section 5.1, i.e., convection (or advection), hydrodynamic dispersion, sorption, and first-order decay (radioactive or biological), are included in this code.

As part of the quality assurance efforts, prior to applying the MT3D code to the PVLF site, the code was first verified with a known analytical solution to ensure that the code could be used to solve the transport equations with sufficient accuracy. The example problem used for this verification is described in detail in Appendix E.3. The MODFLOW code is first used to simulate the flow condition and provide the distribution of ground water flow to the MT3D code which is then used to solve the transport equation. A comparison between the analytical and MT3D solutions using linear scales is shown in Figure 5.2-1. The agreement between the MT3D and analytical solutions is excellent. No numerical oscillation, which is common for problems with Péclet numbers greater than two, is observed. A semi-logarithmic comparison (using the logarithmic scale for the concentration distribution) is presented in Figure 5.2-2, which indicates almost exact agreement between the MT3D and analytical solutions when the variation of concentration remains within two orders of magnitude. As the level of concentration decreases, MT3D tends to overestimate the actual concentration calculated by the analytical solution. For example, MT3D calculates a concentration of 10^{-5} of C_O (original concentration) approximately eighteen feet from the source, while the actual concentration is 10^{-7} of C_O. Based on this observation, the MT3D code may be regarded as conservative at very low concentration levels. Concentration overestimation is due to the effects of truncation errors introduced by the use of finite-difference approximations in the solution procedure

FIGURE 5.2-1

COMPARISON BETWEEN MT3D AND ANALYTICAL SOLUTION, LINEAR SCALE



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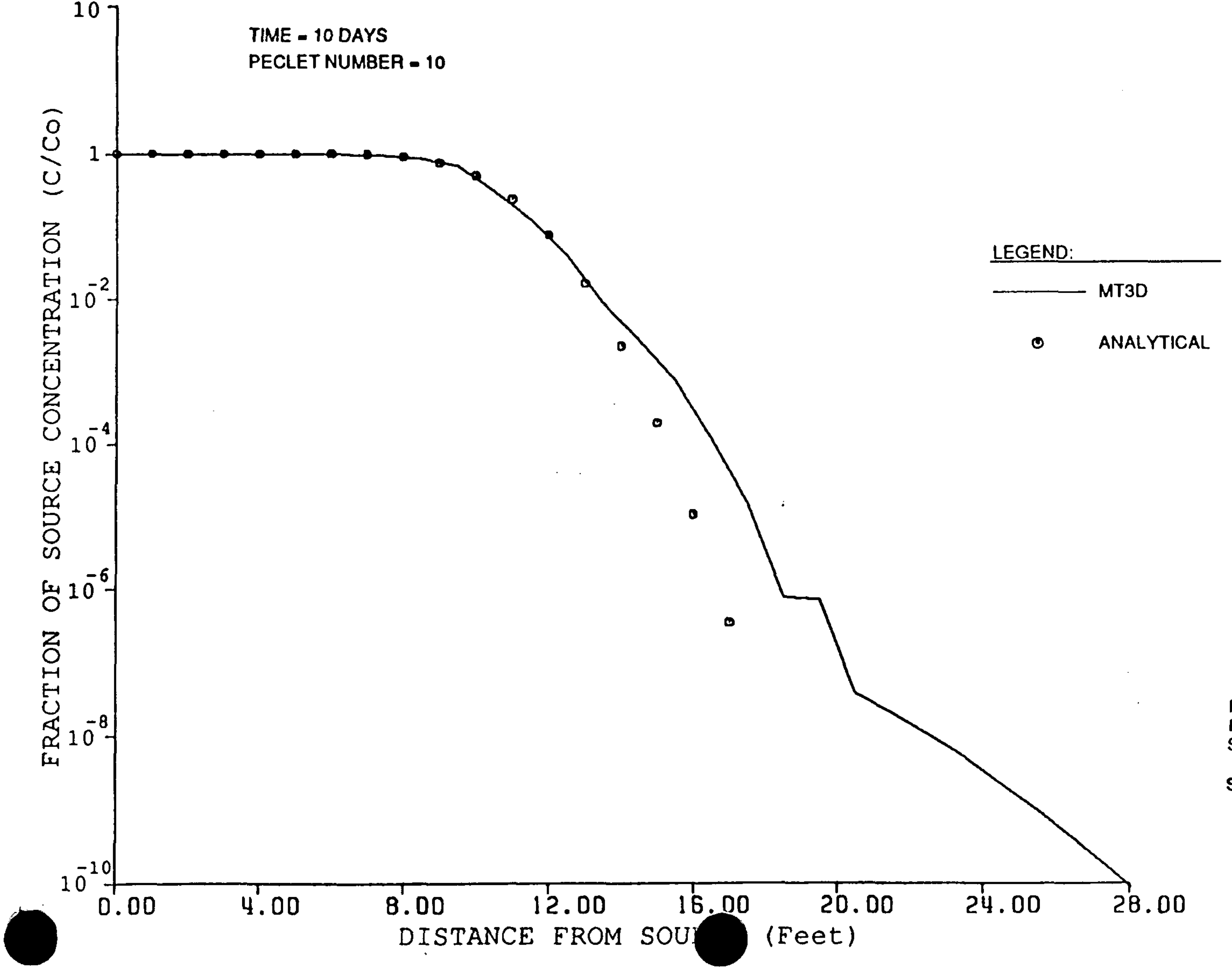
SOURCE: DAMES & MOORE, INC.

FIGURE 5.2-2

COMPARISON BETWEEN MT3D AND ANALYTICAL SOLUTION, SEMI-LOGARITHMIC SCALE



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of the dispersive part of the transport equation. Truncation errors exist in all numerical solutions of the transport equation and cannot be eliminated. They can, however, be reduced by decreasing the grid size at the expense of an increase in computational effort.

5.3 MODEL DEVELOPMENT

In order to simulate three dimensional ground water flow and transport, an areal extent consisting of 6,125 grid blocks covering both upgradient and downgradient of the PVLF was defined. The grid system consists of five layers in the vertical direction, each layer comprising 1,225 (35 x 35) grid blocks. The orientation of the grid was chosen such that one of the principal grid directions is approximately parallel to the trace of the Palos Verdes fault zone downgradient from the PVLF. This gridding arrangement is designed to maximize resolution in the vicinity of the fault immediately downgradient from the PVLF site. Figure 3.4-5 is an example of vertical discretization of the three dimensional model domain. As some grid blocks may contain more than one stratigraphic unit, mathematical averaging techniques need to be applied to define overall hydrogeologic or physico-chemical properties for these grid blocks. The selected averaging methods are described by Gailey (Gailey, et. al., 1991) and are shown below:

Adsorption Property

$$K^{j}{}_{d_{AVE}}^{j} = \sum_{i=1}^{n} K_{di}^{j} \frac{K_i \delta_i}{\sum\limits_{i=1}^{n} K_i \delta_i}$$

Effective Porosity

$$\theta_{AVE} = \frac{\sum\limits_{i=1}^{n} \theta_i K_i \delta_i d_i}{\sum\limits_{i=1}^{n} K_i \delta_i \sum\limits_{i=1}^{n} d_i}$$

where:

K_{dAVE}^{j} = average K_d value of chemical j

K_{di}^{j} = chemical distribution coefficient of formation i for chemical j

K_{di}^{j} = $f_{oc}^{i} \quad K_{oc}^{j}$ for organic chemicals

f_{oc}^{i} = organic carbon fraction of formation i

K_{oc}^{j} = distribution coefficient of organic chemical j

K_i = hydraulic conductivity of formation i

d_i = thickness of formation i within the computational cell of interest

δ_i = Kronecker's delta; equals 0 for $d_i = 0$ and 1 for $d_i > 0$

θ_{AVE} = average effective porosity

θ_i = effective porosity of formation i

Details regarding the averaging of the hydraulic conductivity values are presented in Appendix E.2. Mathematical formulation of the developed ground water contaminant transport model is detailed in Appendix E.3.

5.4 MODEL INPUT

Waste disposal history and waste characteristics have been described in detail earlier in Section 1.3.2 of this report. As shown in Exhibits 1.3-1 and 1.3-2, the landfill is divided into three general areas with three types of disposal areas based on topographic, geologic, and waste characteristics. Because of these differences, the amount of potential chemical discharge would vary according to the area of the landfill. To account for the variability of the discharge of chemicals from all of the landfill parcels, the three landfill classes are divided into the zones shown in Figure 5.4-1.

For the purpose of contaminant transport modeling, the PVLF is divided into six zones (see Figure 5.4-1). Waste disposal history and the type of waste disposed of in each zone are summarized below.

Zone	Classification	Potential discharge history
1	II	From 1970
2a	II	From 1952
2b	II	From 1952
3	II	From 1964
4a	I	From 1961
4b	I	From 1961
5	I (Engineered)	From 1971
6	Injection wells	1975-1985

Zones 2 and 4 are further subdivided into two subzones to account for spatial variability. The injection rates of the two injection wells, GSF-1 and GSF-2, were approximately 2,200 gallons per day. GSF-1 was operated from 1975 to 1984, and GSF-2 from 1984 to July 1985. Approximate locations of these two injection wells are also shown in Figure 5.4-1.

In each zone, it is conservatively assumed chemical discharge began immediately after the disposal of waste. The infiltration flux is that obtained from the calibrated flow model. The concentration of each chemical for each zone is found by matching the model simulation results with the data between 1986 to 1991. The calibrated input chemical concentrations are summarized in Table 5.4-1.

5.5 MODEL PARAMETERS

The ground water contaminant transport model requires the following input parameters:

1. Effective porosity (θ_e);
2. Organic carbon fraction (f_{oc});
3. Distribution coefficient (K_d);

TABLE 5.4-1

CONCENTRATIONS IN DISPOSAL ZONES FROM MODEL CALIBRATION

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Chemical	Zone								Unit
	1	2A	2B	3	4A	4B	5	6 (injection)	
Nitrate	3×10^3	3×10^3	8×10^4	1×10^3	1×10^1	1×10^1	5×10^3	0	mg/L
Vinyl Chloride	1×10^3	1×10^4	1×10^2	1×10^3	1×10^3	1×10^6	1×10^2	1×10^6	μg/L
Benzene	1×10^3	1×10^4	5×10^5	1×10^5	1×10^3	1×10^5	2×10^3	4×10^7	μg/L
Trichloroethylene	4×10^5	8×10^5	1×10^3	1×10^3	1×10^3	1×10^3	1×10^3	1×10^3	μg/L
Tetrachloroethylene	1×10^5	8×10^6	1×10^5	1×10^5	1×10^6	1×10^6	1×10^4	2×10^7	μg/L
Ethyl Benzene	5×10^4	5×10^3	5×10^6	1×10^5	1×10^4	1×10^6	1×10^3	2×10^6	μg/L
Chlordane	1×10^2	1×10^4	5×10^4	1×10^2	1×10^1	5×10^6	0	0	mg/L
Uranium	1×10^7	1×10^7	1×10^7	1×10^7	0	0	0	0	pCi/L
Tritium	0	0	6×10^5	0	0	0	0	0	pCi/L

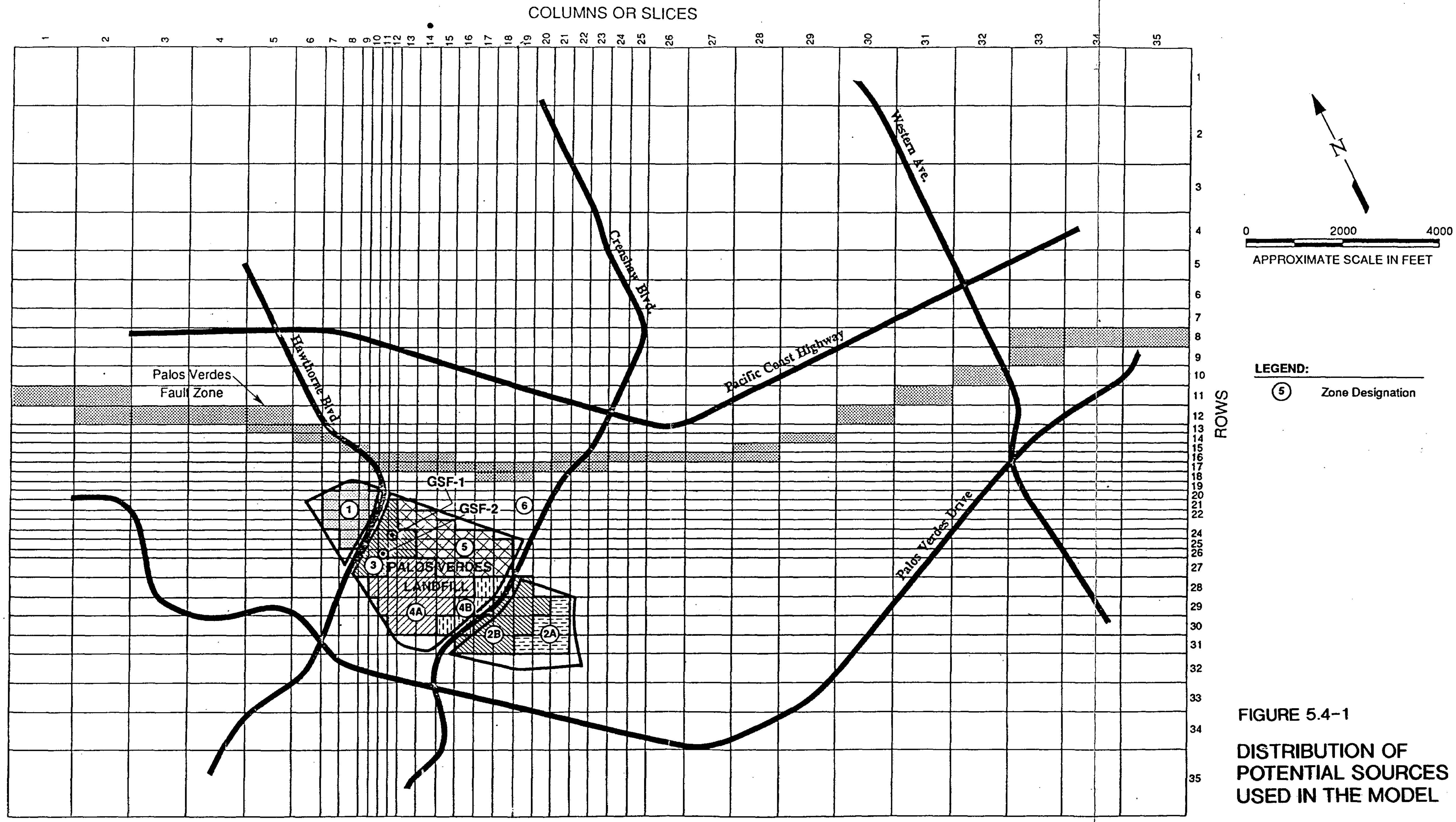

FIGURE 5.4-1

DISTRIBUTION OF POTENTIAL SOURCES USED IN THE MODEL

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SOURCE: DAMES & MOORE, INC.

4. Partition coefficient for solid matrix organic carbon to water (K_{oc});
5. Longitudinal dispersivity (a_L);
6. Lateral dispersivity (a_T);
7. Tortuosity (T);
8. Molecular diffusion coefficient (D_d); and
9. Half life ($t_{½}$).

Definitions of the above parameters are presented in Appendix E.3. These parameters are discussed in the following paragraphs, and the values assumed for the transport parameters are summarized in Table 5.5-1.

Average total porosity values obtained by Herzog and Associates (1991a, b) for fill, alluvium/colluvium, San Pedro sand/Lomita marl, Malaga Mudstone, Valmonte Diatomite, and Altamira Shale, are, respectively, 0.46, 0.45, 0.44, 0.58, 0.53, and 0.45. It should be noted that these values were determined in the laboratory and represent the magnitude of total porosity (total void space including dead-end pores), not effective porosity. In the Monterey Formation, the measured porosities are most likely the primary porosity (porosity of the solid matrix). The Monterey Formation comprises fractured mudstone, diatomite, and shale in which the majority of flow and transport occurs in the secondary porosity (porosity which is structurally controlled). Since the secondary porosities of the geologic units within the Malaga Mudstone member are not available, ranges published by Driscoll (1986) for effective porosity are presented in Table 5.5-1. These effective porosity values are one order of magnitude smaller than the Herzog values listed and were employed as an initial estimate of the secondary porosity of the Monterey Formation. No field or laboratory data are available for effective porosity of the sedimentary deposits overlying the Monterey Formation. However, the values of effective porosity in porous media (sedimentary deposits) are normally smaller than those of the total porosity (Bear, 1972). In this study, the values published by Driscoll (1986), which are appropriately smaller than those reported by Herzog and Associates (1991a, b), were adopted.

The other transport parameters were obtained from the literature. A comprehensive review of dispersivity values may be found in Pickens and Grisak (1981). In this work, longitudinal dispersivity values obtained from modeling of large-scale ground water contamination in several

TABLE 5.5-1

TRANSPORT MODEL PARAMETERS

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Parameter	Formation	Value
f_{oc} (Organic carbon fraction)	Fill	1×10^{-3} (1)
	Alluvium	15×10^{-3} (2)
	San Pedro Sand	3×10^{-3} (1)
	Malaga Mudstone	67.4×10^{-3} (2)
	Valmonte Diatomite	52.4×10^{-3} (2)
	Altamira Shale	12.4×10^{-3} (2)
	Catalina Schist	1.5×10^{-3} (1)
θ_e (Effective porosity)	Fill	0.33 (3)
	Alluvium	0.33 (3)
	San Pedro Sand	0.33 (3)
	Malaga Mudstone	0.03 (3)
	Valmonte Diatomite	0.1 (3)
	Altamira Shale	0.03 (3)
	Catalina Schist	0.03 (3)
a_L (Longitudinal dispersivity)	All	50 ft (4)
a_T/a_L (Longitudinal/ lateral dispersivity ratio)	All	0.1 (5)
T (Tortuosity)	All	0.7 (6)
D_d (Molecular diffusion coefficient)	All	1×10^{-9} m^2/s (7)(8)

Notes:
(1) Estimated from similar materials.
(2) Analytical results obtained by the Sanitaion Districts.
(3) Estimated from the range published by Driscoll (1986).
(4) Value reported by Pickens and Grisak (1981).
(5) Common ratio cited by de Marsily (1986).
(6) Value for unconsolidated materials (Bear, 1972).
(7) Freeze and Cherry (1979).
(8) Barone, et al. (1992).

alluvial aquifers varied between twelve meters (39 feet) and 61 meters (200 feet). Data from a two-well tracer test in an alluvial aquifer in Barstow, California (Robson, 1974), indicated a dispersivity value range of 50 to 100 feet. These values have, however, a scale-dependent tendency (i.e., the larger the scale, the larger the dispersivity value). Only limited information relating to dispersivity in fractured media is available in literature. Longitudinal dispersivity from Grove and Beteem's (1971) tracer tests in a fractured carbonate aquifer is approximately 125 feet. In the PVLF study, a longitudinal dispersivity value range of 50 to 100 feet is assumed for all formations. Note that the reported ranges of dispersivity in sedimentary deposits and fractured rock mass are of similar values.

The ratio of lateral dispersivity to longitudinal dispersivity is known to vary between 1/5 to 1/100 (de Marsily, 1986). Pickens and Grisak (1981) reported a range of 1/3 to 1/5 in areal modeling of ground water contamination. These ratios were obtained through a calibration process. The greater the ratio, the more lateral mixing would occur, and the lower the concentration in the vicinity of the front. In this study, a ratio of 1/10 is assumed. This is the ratio commonly found in modeling studies of ground water contamination (Pinder, 1977). This ratio is approximately one third of the maximum value of the reported range and is therefore considered conservative. The smaller the ratio, the greater the concentration along the center line of the chemical plume.

The value of tortuosity normally ranges between 0.3 and 0.7 (Bear, 1972). Values as low as 0.15 to 0.2 have been found in experiments with mudstone samples from Oklahoma (Barone, et al., 1992). The high end of the range is applicable to unconsolidated materials and the low end to consolidated materials (Bear, 1972). In this study, the upper bound value is assumed since the majority of chemical transport occurs in the alluvium and San Pedro sand. This value is conservatively applied to all hydrogeologic units.

Ranges of anaerobic and ground water biodegradation half-lives ($t_{1/2}$), typical values of partition coefficients (K_{oc}) for organic chemicals, and distribution coefficients (K_d) for inorganic chemicals are summarized in Table 5.5-2. These values were obtained from published literature (Howard, et al. 1991; Fetter, 1988; Montgomery and Welkom, 1990; Strenge and Peterson, 1989; and Verschueren, 1990). Four categories of compounds are listed in Table 5.5-2. These categories are: (1) inorganic chemicals; (2) organic chemicals including both volatile and semi-volatile compounds; (3) radionuclides; and (4) metals. Although both aerobic and anaerobic biodegradation processes

TABLE 5.5-2

LIST OF COMPOUNDS AND THEIR HALF-LIFE, PARTITION COEFFICIENT (K_{OC}), AND DISTRIBUTION COEFFICIENT (K_d)

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COMPOUND	GROUNDWATER BIODEGRADATION HALF-LIFE (years)	ANAEROBIC BIODEGRADATION HALF-LIFE (years)	RADIOACTIVE HALF-LIFE (years)	K_{OC}	K_d
INORGANIC CHEMICALS					
AMMONIA NITROGEN	NA	NA	NA	NA	0
NITRATE NITROGEN	NA	NA	NA	NA	0
ORGANIC COMPOUNDS					
ACETONE	0.01-0.04	0.01-0.08	NA	1	NA
BENZENE	0.02-0.66	0.02-0.06	NA	49	NA
BROMODICHLOROMETHANE	Not Available	Not Available	NA	131	NA
BROMOFORM	0.15-1.00	0.31-2.00	NA	281	NA
CARBON TETRACHLORIDE	0.02-1.00	0.02-0.08	NA	232	NA
CIS-CHLORDANE	1.55-7.60	0.003-0.02	NA	53,000	NA
TECHNICAL CHLORDANE	1.55-7.60	0.003-0.02	NA	532,000	NA
TRANS-CHLORDANE	1.55-7.60	0.003-0.02	NA	1,000,000	NA
CHLOROBENZENE	0.37-0.82	0.75-1.64	NA	318	NA
BIS(2-CHLOROETHYL)ETHER	0.15-1.00	0.31-2.00	NA	14.1	NA
CHLOROFORM	0.15-5.00	0.02-0.08	NA	34	NA
CHLOROMETHANE (methyl chloride)	0.04-0.15	0.08-0.31	NA	43	NA
2-CHLOROPHENOL	Not Available	Not Available	NA	27	NA
PP'-DDD	0.19-31.30	0.19-0.81	NA	238,000	NA
PP'-DDE	0.04-31.30	0.04-0.27	NA	155,000	NA
1,4-DICHLOROBENZENE	0.15-1.00	0.31-2.00	NA	594	NA
1,1-DICHLOROETHANE	0.18-0.42	0.35-1.69	NA	45	NA
1,2-DICHLOROETHANE	0.27-1.00	1.10-2.00	NA	36	NA
1,1-DICHLOROETHENE	0.15-0.36	0.22-0.47	NA	217	NA
CIS-1,2-DICHLOROETHENE	0.15-7.88	0.31-2.00	NA	59	NA
TRANS-1,2-DICHLOROETHENE	0.15-7.88	0.31-2.00	NA	59	NA
1,2-DICHLOROPROPANE	0.92-7.10	1.83-14.10	NA	51	NA
CIS-1,3-DICHLOROPROPENE	0.02-0.03	0.08-0.31	NA	48	NA
TRANS-1,3-DICHLOROPROPENE	0.02-0.03	0.08-0.31	NA	48	NA
DIELDRIN	0.003-6.00	0.003-0.02	NA	25,120	NA
DIETHYL PHTHALATE	0.02-0.31	0.08-0.61	NA	69	NA
DI-N-BUTYL PHTHALATE	0.04-1.00	0.50-1.00	NA	217	NA
ETHYL BENZENE	0.02-0.62	0.48-0.62	NA	622	NA
ISOPHORONE	0.04-0.15	0.08-0.31	NA	26	NA
METHYLENE CHLORIDE (dichloromethane)	0.04-0.15	0.08-0.31	NA	8.7	NA
N-NITROSODI-N-PROPYLAMINE	0.12-1.00	0.23-2.00	NA	31	NA

TABLE 5.5-2 (CONTINUED)
LIST OF COMPOUNDS AND THEIR HALF-LIFE, PARTITION COEFFICIENT (K_{OC}), AND DISTRIBUTION COEFFICIENT (K_d)
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COMPOUND	GROUNDWATER BIODEGRADATION HALF-LIFE (years)	ANAEROBIC BIODEGRADATION HALF-LIFE (years)	RADIOACTIVE HALF-LIFE (years)	K_{oc}	K_d
N-NITROSODIPHENYLAMINE	0.05-0.19	0.11-0.37	NA	982	NA
PENTACHLOROPHENOL	0.13-4.20	0.12-4.20	NA	900	NA
PHENOL	0.001-0.02	0.02-0.08	NA	27	NA
TETRACHLOROETHYLENE	1.00-2.00	027-4.50	NA	267	NA
TETRAHYDROFURAN	Not Available	Not Available	NA	1.8	NA
TOLUENE	0.02-0.08	0.15-0.58	NA	242	NA
1,1,2-TRICHLOROETHANE	0.37-2.00	2.00-4.00	NA	49	NA
TRICHLOROETHYLENE	0.88-4.50	0.27-4.50	NA	97	NA
VINYL CHLORIDE	0.15-7.88	0.31-2.00	NA	2.5	NA
M+P-XYLENE	0.04-1.00	0.50-1.00	NA	550	NA
RADIOACTIVE CHEMICALS					
GROSS ALPHA RADIOACTIVITY	NA	NA	NA	NA	NA
TRITIUM	NA	NA	12.3	NA	0
URANIUM 224	NA	NA	244,000	NA	50
URANIUM 228	NA	NA	4,470,000,000	NA	50
RADIUM 226+228	NA	NA	1600	NA	24.3
METALS					
ARSENIC	NA	NA	NA	NA	5.86
BARIUM	NA	NA	NA	NA	530
BORON	NA	NA	NA	NA	1.3
CADMIUM	NA	NA	NA	NA	14.9
HEXAVALENT CHROMIUM	NA	NA	NA	NA	16.5
TOTAL CHROMIUM	NA	NA	NA	NA	16.5
COPPER	NA	NA	NA	NA	41.9
LEAD	NA	NA	NA	NA	234
MANGANESE	NA	NA	NA	NA	16.5
MERCURY	NA	NA	NA	NA	322
NICKEL	NA	NA	NA	NA	12.2
SELENIUM	NA	NA	NA	NA	5.91
SILVER	NA	NA	NA	NA	0.4
ZINC	NA	NA	NA	NA	12.9

Note: NA = Not Applicable

occur in ground water, only the half-lives for the anaerobic biodegradation are listed in Table 5.5-2. Anaerobic biodegradation generally takes longer than aerobic degradation and is therefore more critical, and more conservative, for the purpose of estimating risks. To account for field conditions, the ranges for biodegradation half-lives in ground water (Howard, et. al., 1991) are also listed on Table 5.5-2 for comparison.

As shown in Table 5.5-2, the range of biodegradation half-life of each chemical varies from a fraction of a year to several years. Such a wide range occurs because of variability of related factors such as concentration of dissolved oxygen, available nutrients, and density of microorganisms. The maximum biodegradational half-lives for most chemicals are well below fourteen years, except for pp'-DDD and pp'-DDE (31 years for both compounds).

In addition to biodegradation, abiotic transformation of chemicals also occurs in natural ground water environments as described by Vogel and McCarty (1987). The developed contaminant transport model includes abiotic degradation as a first order decay process. This allows for the combination of abiotic transformation and biodegradation of organic compounds because both processes are described by a first order kinetic model. Vogel and McCarty reported the abiotic degradation half-life for 1,1,1-trichloroethane ranges from 2.8 to 17 years in ground water environment. To simplify the sensitivity analysis and to account for the uncertainties associated with the wide ranges of parameter values, an extremely conservative assumption is made in this study that the half-life for all modeled organic compounds is 17 years. This assumption is conservative because the assumed long half life translates to a very low degradation rate for the modeled compounds. As stated above, only two modeled compounds, pp'-DDD and pp'-DDE, have longer half lives than 17 years. Since both pp'-DDD and pp'-DDE are relatively immobile (their K_{oc} values being greater than 100,000) and they have been rarely detected at site monitoring wells, the assumption of a 17 year half life for these compounds is not expected to significantly affect the modeled pp'-DDD and pp'-DDE concentrations.

5.6 MODEL ASSUMPTIONS

The following assumptions have been adopted in the development of the PVLF ground water contaminant transport model.

(1) The flow is steady-state. The justification of this assumption has been discussed in detail in Appendix E.2.

(2) Loss of chemicals from the ground water due to volatilization is negligible. This assumption is conservative for chemicals which may volatilize (e.g., trichloroethylene) when considering only the ground water.

(3) Potential increase of VOC concentration due to landfill gas dissolution into ground water is not considered. Landfill gas generally contains trace levels of volatile organic compounds which may dissolve into ground water upon contact. This process is not considered in the development of the contaminant transport model because it is relatively insignificant to the assumed source of VOCs from the landfill. In addition, the concentrations of VOCs in landfill gas generally decrease with time.

(4) Density effects are negligible. This assumption is justified by the fact that the sum of average concentrations of dissolved chemicals at the site is below one percent of the water density (de Marsily, 1986).

(5) Decay parameters (biodegradational, abiotic-transformational, and radioactive half-lives) are spatially uniform. This assumption is particularly true for radioactive materials; however, for organic chemicals the half-lives are likely to be spatially variable because of the variability of related attributes, such as the density of microorganisms and availability of nutrients. To circumvent this uncertainty, an extremely conservative half-life value of 17 years for organic chemicals was adopted in this study as explained in Section 5.5. Note that the larger the half-life, the longer the chemical remains in the flow system, and the larger the chemical concentration at potential receptors.

(6) Generation of organic compounds as a result of degradation of parent compounds (e.g., degradation of tetrachloroethylene produces trichloroethylene) is not considered in this study. This assumption allows independent modeling of individual chemicals, which simplifies the mathematical formulation of the model and results in

manageable solution of the model. The omission of gain due to chemical transformation is conservatively compensated by the assumption of a 17 year half life for degradation of all organic compounds.

(7) Adsorptive parameters (organic carbon fraction, f_{oc}, and distribution coefficient K_d) are spatially uniform in each formation. In the Monterey Formation, the relationship between f_{oc} and the retardation factor is not exactly known. In this formation, the convective component (transport of chemicals due to moving ground water) occurs mainly in secondary porosity. Other transport processes, such as matrix diffusion along intergranular or intragranular microfissures in the background solid matrix, could also occur and could play a significant role in retarding the advance of the chemical front. Similarly, the distribution coefficient, K_d, may also be spatially dependent because of the variability of soil composition (e.g., percentage of fine-grained materials). This uncertainty is circumvented by assuming that the adsorptive properties of the geologic media within the model area are a fraction (1/10 to 1/2) of those determined from the f_{oc} and K_d values reported in Tables 5.5-1 and 5.5-2.

(8) Each chemical of concern originates from sources within the PVLF. It is further assumed that these sources are continuous and uniform in time. This assumption implies that concentration of each chemical of concern observed in ground water monitoring wells in the vicinity of and downgradient from the landfill is due to off site migration from the sources within the landfill. This assumption is, again, extremely conservative because the strengths of sources within the PVLF (if they exist) are likely to diminish with time due to chemical and biological degradation. In addition, a review of ground water monitoring data (see Section 3.6) indicates that there is no evidence of leachate generation and migration from the PVLF. Furthermore, as described in Section 1.3.4.3.1, a subsurface barrier system downgradient of the main site along Hawthorne Boulevard has been present since 1986. Any ground water flow in the alluvium is collected by a system of extraction wells upgradient of the subsurface barrier and is treated before being sewered under an existing Industrial Waste permit.

9) The spatial distribution of each chemical concentration in 1986 is representable by the spatial distribution of the maximum concentration of each respective chemical monitored between 1986 and 1991. The assumed distribution of concentration of each principal chemical is utilized to provide a conservative estimate of the strengths of potential sources of that principal chemical within the PVLF. Again, this is a very conservative assumption made for the purpose of baseline risk assessment and to account for parameter uncertainty. As discussed in Section 3.6, statistically significant decreasing trends have been observed for a few water quality parameters at monitoring wells downgradient of the PVLF.

As can be seen, the above assumptions are designed to result in conservative estimates of modeled chemicals such that the model results can be used for evaluation of health risks associated with the ground water transport pathway based on the maximum expected concentrations.

5.7 CALIBRATION OF THE TRANSPORT MODEL

Prior to calibration of the transport model, the contaminants to be modeled were determined. Calibration objectives were determined, followed by calibration modeling. The following sections discuss the modeled contaminants, calibration modeling, and the results of the calibration modeling.

5.7.1 Modeled Contaminants

Because the estimate of concentrations at potential receptors of a large number of potential contaminants was required for the baseline risk assessment, the contaminants of concern were divided into subgroups in accordance with their adsorptive properties since the transport depends primarily on these properties (all other mechanisms being equal). In each subgroup, a principal chemical was chosen for detailed calibration and predictive analysis. Concentrations of other chemicals within the subgroup at potential receptors may be estimated by scaling the breakthrough curves of the principal chemical at potential receptors.

The chemicals are listed in Table 5.7-1 according to groups, subgroups, and the order of their respective adsorptive properties. It should be noted that the K_d values are listed for inorganic chemicals and K_{oc} values for organic chemicals. The chemical groups are discussed below.

5.7.1.1 Inorganic Chemicals

Nitrate was chosen as a conservative chemical to represent this group because of its mobility and the fact that it does not biodegrade. Nitrate is used to verify and calibrate parameters relating to ground water velocity and hydrodynamic dispersion. For this purpose, conservative chemicals are normally used. In this study, two conservative chemicals are considered: chloride and nitrate. Nitrate, however, is preferred over chloride because of the following reasons:

- The presence of nitrate is most likely to be due to anthropogenic activities, whereas the presence of chloride may be due to natural causes such as saltwater intrusion, the presence of marine sediments, etc.
- Chloride concentration data are associated with highly variable background concentrations, and are difficult to use for model calibration.

5.7.1.2 Organic Chemicals

This group of chemicals was divided into six subgroups according to K_{oc} values. A principal chemical was selected in each subgroup. These subgroups are listed below, in ascending order of K_{oc} values.

Subgroup 1 (K_{oc}: 1 to 14.1)

Vinyl chloride was chosen as the principal chemical for this subgroup, which is the most mobile of the six subgroups.

TABLE 5.7-1
PARTITION COEFFICIENT (K_{OC}) AND DISTRIBUTION COEFFICIENT (K_d) VALUES
PALOS VERDES LANDFILL - REMEDIAL INVESTIGATION REPORT

COMPOUND	K_{OC}	K_d
INORGANIC CHEMICALS		
AMMONIA NITROGEN	NA	0
NITRATE NITROGEN	NA	0
ORGANIC COMPOUNDS		
SUBGROUP 1		
ACETONE	1	NA
TETRAHYDROFURAN	1.8	NA
VINYL CHLORIDE	2.5	NA
METHYLENE CHLORIDE (dichloromethane)	8.7	NA
BIS(2-CHLOROETHYL)ETHER	14.1	NA
SUBGROUP 2		
ISOPHORONE	26	NA
PHENOL	27	NA
2-CHLOROPHENOL	27	NA
N-NITROSODI-N-PROPYLAMINE	31	NA
CHLOROFORM	34	NA
1,2-DICHLOROETHANE	36	NA
CHLOROMETHANE (methyl chloride)	43	NA
1,1-DICHLOROETHANE	45	NA
TRANS-1,3-DICHLOROPROPENE	48	NA
CIS-1,3-DICHLOROPROPENE	48	NA
1,1,2-TRICHLOROETHANE	49	NA
BENZENE	49	NA
1,2-DICHLOROPROPANE	51	NA
TRANS-1,2-DICHLOROETHENE	59	NA
CIS-1,2-DICHLOROETHENE	59	NA
DIETHYL PHTHALATE	69	NA
SUBGROUP 3		
TRICHLOROETHYLENE	97	NA
BROMODICHLOROMETHANE	131	NA
SUBGROUP 4		
1,1-DICHLOROETHENE	217	NA
DI-N-BUTYL PHTHALATE	217	NA
CARBON TETRACHLORIDE	232	NA
TOLUENE	242	NA
TETRACHLOROETHYLENE	267	NA
BROMOFORM	281	NA
CHLOROBENZENE	318	NA

TABLE 5.7-1 (CONTINUED)
PARTITION COEFICIENT (K_{OC}) AND DISTRIBUTION COEFICIENT (K_D) VALUES
PALOS VERDES LANDFILL - REMEDIAL INVESTIGATION REPORT

COMPOUND	K_{OC}	K_d
SUBGROUP 5		
M+P-XYLENE	550	NA
1,4-DICHLOROBENZENE	594	NA
ETHYL BENZENE	622	NA
PENTACHLOROPHENOL	900	NA
N-NITROSODIPHENYLAMINE	982	NA
SUBGROUP 6		
DIELDRIN	25,120	NA
CIS-CHLORDANE	53,000	NA
PP'-DDE	155,000	NA
PP'-DDD	238,000	NA
TECHNICAL CHLORDANE	532,000	NA
TRANS-CHLORDANE	1,000,000	NA
RADIOACTIVE CHEMICALS		
TRITIUM	NA	0
RADIUM 226+228	NA	24.3
URANIUM 224	NA	50
URANIUM 228	NA	50
METALS		
SILVER	NA	0.4
BORON	NA	1.3
ARSENIC	NA	5.86
SELENIUM	NA	5.91
NICKEL	NA	12.2
ZINC	NA	12.9
CADMIUM	NA	14.9
MANGANESE	NA	16.5
TOTAL CHROMIUM	NA	16.5
HEXAVALENT CHROMIUM	NA	16.5
COPPER	NA	41.9
LEAD	NA	234
MERCURY	NA	322
BARIUM	NA	530

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Notes:

Vinyl Chloride is the principal chemical for Subgroup 1 of organic compounds.
Benzene is the principal chemical for Subgroup 2 of organic compounds.
Trichloroethylene is the principal chemical for Subgroup 3 of organic compounds.
Tetrachloroethylene is the principal chemical for Subgroup 4 of organic compounds.
Ethyl benzene is the principal chemical for Subgroup 5 of organic compounds.
Chlordane is the principal chemical for Subgroup 6 of organic compounds.
NA=Not Applicable

Subgroup 2 (K_{oc}: 26 to 69)

Benzene was chosen as the principal chemical for this subgroup.

Subgroup 3 (K_{oc}: 97 to 131)

Trichloroethylene was chosen as the principal chemical for this subgroup.

Subgroup 4 (K_{oc}: 217 to 318)

Tetrachloroethylene was chosen as the principal chemical for this subgroup.

Subgroup 5 (K_{oc}: 550 to 982)

Ethyl benzene was chosen as the principal chemical for this subgroup.

Subgroup 6 (K_{oc}: 25,120 to 1,000,000)

Cis-chlordane was chosen as the principal chemical for this subgroup. In this study, cis-chlordane is referred to as simply chlordane.

5.7.1.3 Radionuclides

Two radionuclides were chosen as principal chemicals: tritium and uranium. Radium is not considered for the analysis because of its relatively low concentration. It should be noted that the analysis of the two principal radionuclides is based on a limited amount of available field data.

5.7.1.4 Metals

The available monitoring results for metals (see Section 3.6.2.2 for a discussion of metal concentrations at site monitoring wells) indicate a high degree of uncertainty. For example, the total chromium concentrations within the local flow regime (i.e., landfill area, downgradient from the landfill and adjacent to the landfill) and the regional flow regime (West Coast Basin) are of the same order of magnitude. This observation could be attributable to highly variable background concentrations due to natural mineral sources and/or the use of unfiltered samples for metal analysis resulting in interference from suspended solids in the ground water.

The observed chromium levels are inconsistent with the fact that the K_d value of chromium is relatively high, rendering chromium relatively immobile. Until a database for field filtered metals concentration is established, the available data may not be reliably used to calibrate the transport model. The Sanitation Districts began analyzing filtered samples for metals concentration in ground water in January 1993.

Most metals present in the ground water at the PVLF are relatively immobile because of their high K_d values. It is considered unlikely for metals to travel a great distance from the landfill. Because of the data uncertainty and relative immobility of most metals, they are not explicitly modeled in this study. However, there are two metals, silver and boron, that are relatively mobile as indicated by their low K_d values. The transport characteristics of these two metals are therefore similar to those of nitrate. For modeling purposes, concentrations of these two metals at potential receptors may be scaled from those of nitrate.

5.7.2 Calibration Modeling Objectives

The flow and transport models were calibrated by the trial-and-error method to achieve the following calibration objectives:

- Agreement between the simulated ground water piezometric head (ground water level) and the observed ground water levels. This step was conducted to ensure that the flow model provides a reliable ground water velocity distribution for the transport model.

- Agreement between the simulated locations of chemical fronts based on the known disposal history and the available chemical data between 1986 and 1991. This step provides an additional step of flow model verifications and ensures that the transport model can accurately simulate the movement of chemical fronts.

- Agreement between the simulated concentration distribution and the observed distribution of chemical concentrations based on available field data between 1986 and 1991. This step ensures that the transport model can simulate the major transport characteristics of each chemical evaluated.

For the ground water flow model calibration, a total of 43 monitoring wells were used and a correlation coefficient of 0.98 between the modeled and observed hydraulic heads was obtained. The high degree of correlation indicates that the ground water flow model adequately describes the hydraulic head distribution in the modeled domain. The ground water velocity distribution is obtained from the calibrated flow model for use in the contaminant transport model.

For calibration of the contaminant transport model, the maximum concentration of the modeled parameter at each monitoring well between 1986 and 1991 was used for conservatism. For each calibration run, the simulation is terminated in 1986 prior to the installation of the barrier along Hawthorne Boulevard. The barrier is included for predictive analysis runs, with each run beginning in 1986. The reasons for using 1986 as the cut-off point in time are: (1) the flow pattern in the vicinity of the barrier was modified after the installation of the barrier; and (2) the ground water monitoring program for the PVLF site began in 1986. The barrier is incorporated into the model by assigning a hydraulic conductivity value of 10^{-7} cm/sec, which is typical for cement/bentonite, to blocks representing the barrier.

5.7.3 Calibration Modeling Results

Calibration of the contaminant transport model was carried out for nitrate, vinyl chloride, benzene, trichloroethylene, tetrachloroethylene, ethyl benzene, chlordane, uranium, and tritium. Monitoring results from 44 wells downgradient of the PVLF were used for the calibration because concentrations of chemicals at upgradient wells represent background conditions, not chemicals potentially generated from the PVLF. Results from the model calibration and concentration profiles calculated from model calibration are detailed in Appendix E.3.

Two statistical parameters are used to measure the goodness of fit between the observed and modeled concentrations. These two measures are the correlation coefficient and the geometric mean concentrations of the modeled chemicals. The correlation coefficient is unity when the data are perfectly correlated. The critical correlation coefficient is 0.393 for the contaminant transport model calibration at a level of significance of one percent with 42 degrees of freedom. In other words, the correlation between modeled and observed concentrations is statistically significant if the correlation coefficient is greater than 0.393. The observed and modeled geometric mean concentrations for a chemical indicate positive predictive bias, i.e., degree of over-estimation of modeled concentration. Results of contaminant transport model calibration for the principal chemicals are summarized in Table 5.7-2 and are briefly discussed below.

5.7.3.1 Nitrate

Nitrate is first calibrated because the effects due to biodegradation and adsorption can be isolated (i.e., nitrate is chemically inert and relatively mobile in ground water). By the trial-and-error method, the combination of nitrate concentration for each zone was estimated by matching the model results with the maximum nitrate concentration between 1986 and 1991. The calibrated nitrate discharge concentrations for all the zones are presented in Table 5.4-1. Table 5.7-2 indicates that the agreement between the modeled and observed nitrate concentrations is favorable except for Case 2 where modeled concentrations in 1991 are compared with the average concentrations from 1986 to 1991. This is expected since the model calibration is based on the maximum concentrations from 1986 to 1991. Correlations between the modeled concentrations for 1986 and 1991 and the maximum concentrations from 1986 to 1991 (Case 1 and Case 3, respectively) are statistically significant (correlation coefficient greater than 0.393). Downgradient from the landfill, north of the PVLF, an anomalous distribution of nitrate concentration is observed in the monitoring database, especially at M26A and M50B across the Palos Verdes fault zone. The nitrate observed in this area could originate from the nearby residential areas (e.g., from fertilization, etc.). No attempt is made to match the model to anomalous concentrations.

TABLE 5.7-2

STATISTICS OF CHEMICAL TRANSPORT SIMULATION

PALOS VERDES LANDFILL - REMEDIAL INVESTIGATION REPORT

Chemical	Case	Standard deviation of residuals (A)	Absolute maximum residual (B)	Correlation coefficient	Geometric average of observed concentration	Geometric average of simulated concentration	Units
Nitrate	1	0.93	2.56	0.466	3.85	3.03	mg/l
	2	1.22	2.17	0.272	0.81	3.32	mg/l
	3	0.93	2.56	0.438	3.85	3.32	mg/l
Vinyl Chloride	1	1.59	3.84	0.831	6.95	77.1	μg/l
	2	1.98	3.88	0.818	2.82	107	μg/l
	3	1.64	3.87	0.835	6.95	107	μg/l
Benzene	1	1.38	2.90	0.570	2.06	5.92	μg/l
	2	1.75	3.80	0.507	0.56	8.04	μg/l
	3	1.36	2.89	0.600	2.06	8.04	μg/l
Trichloroethylene	1	1.09	3.69	0.788	3.55	11.1	μg/l
	2	1.38	3.92	0.759	1.68	14.8	μg/l
	3	1.11	3.73	0.807	3.55	14.8	μg/l
Tetrachloroethylene	1	1.29	4.42	0.489	1.03	1.60	μg/l
	2	1.44	4.42	0.486	0.404	1.87	μg/l
	3	1.33	4.42	0.476	1.0	1.87	μg/l
Ethyl Benzene	1	1.66	3.90	0.503	0.337	4.23	μg/l
	2	2.22	3.83	0.301	0.101	6.86	μg/l
	3	1.79	3.83	0.452	0.337	6.86	μg/l
Chlordane	1	0.153	0.668	0.425	0.052	0.057	μg/l
	2	0.143	0.690	0.425	0.052	0.057	μg/l
	3	0.156	0.690	0.425	0.052	0.057	μg/l
Uranium	1	0.939	3.57	0.407	0.099	0.108	pCi/l
	2	2.29	4.40	0.254	0.098	5.29	pCi/l
	3	2.29	4.40	0.251	0.099	5.29	pCi/l
Tritium	1	0.871	2.67	0.568	0.062	0.160	pCi/l
	2	1.62	2.79	0.381	0.061	1.14	pCi/l
	3	1.62	2.79	0.381	0.062	1.14	pCi/l

Notes:

Case 1) Simulated concentration in 1986 vs. maximum concentration (1986 - 1991).
Case 2) Simulated concentration in 1991 vs. mean concentration (1986 - 1991).
Case 3) Simulated concentration in 1991 vs. maximum concentration (1986 - 1991).

A) Standard deviation of residual of log (concentration).
B) Absolute maximum residual of log (concentration).

5.7.3.2 Vinyl Chloride

By the trial-and-error method, the combination of concentration for each zone was estimated by matching the model with the maximum vinyl chloride concentration between 1986 to 1991. The calibrated vinyl chloride discharge concentrations for all the zones are summarized in Table 5.4-1. Table 5.7-2 indicates the correlation between the modeled and observed vinyl chloride concentrations is statistically significant.

During the course of calibration, it was found that it was necessary to modify the organic carbon fraction (f_{oc}), especially in the alluvium and the Monterey Formation. In the alluvium, the modified f_{oc} is approximately 33 percent smaller than the field value. The values of f_{oc} in the Monterey Formation, however, are up to 85 percent smaller than the field values for the upper two strata, the Malaga Mudstone member and the Valmonte Diatomite member. This could be due to the fact that the measured f_{oc} values reflect the materials taken from the portion predominantly associated with the primary porosity (matrix porosity). Because the major flow and convective transport may be occurring mainly in the secondary porosity (fracture porosity) which has less surface area available for adsorption, the f_{oc} values had to be adjusted accordingly. The original values and the calibrated values of f_{oc} are presented below. It is found that further adjustments of f_{oc} do not significantly improve calibration results. For example, the calibrated f_{oc} values can be further reduced by a factor of two or three (shown in the last column below), but the vinyl chloride concentration distribution obtained for 1986 remains similar to that with the calibrated f_{oc} shown in the second column below. The calibrated discharge rates for vinyl chloride were modified during this calibration.

Formation	Original f_{oc}	Calibrated f_{oc}	Second Alternative f_{oc}
Qo (Fill)	1.0e-03[1]	1.0e-03	0.5e-03
Qo (Alluvium)	15.5e-03	10.5e-03	3.5e-03
Qus	3.0e-03[1]	3.0e-03	1.0e-03
Tmm	67.4e-03	10.5e-03	3.5e-03
Tmv	52.4e-03	10.5e-03	3.5e-03
Tma	12.4e-03	7.2e-03	2.4e-03
Jc	1.5e-03[1]	1.5e-03	0.5e-03

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[1] Data not available, values were assumed based on experience with similar materials in Southern California.

Subsequent sensitivity analysis indicates that the model is very sensitive to the uncertainty associated with the organic carbon fraction when the vinyl chloride discharge rates are held constant. The decrease in the value of f_{oc} causes concentrations at potential receptors to increase. The second alternative set of f_{oc} listed in the last column above was therefore used for risk assessment and as the basis for the calibration of subsequent organic chemicals.

5.7.3.3 Benzene

By the trial-and-error method, the combination of concentration for each zone was estimated by matching the model with the maximum benzene concentration between 1986 and 1991. The calibrated benzene discharge concentrations for all the zones are summarized in Table 5.4-1. The agreement between the modeled and observed benzene concentrations is statistically significant as indicated in Table 5.7-2.

5.7.3.4 Trichloroethylene (TCE)

By the trial-and-error method, the combination of concentration for each zone was estimated by matching the model with the maximum TCE concentration between 1986 and 1991. The calibrated TCE discharge concentrations for all the zones are summarized in Table 5.4-1. The agreement between the modeled and observed TCE concentrations is statistically significant as indicated in Table 5.7-2.

5.7.3.5 Tetrachloroethylene (PCE)

By trial-and-error, the combination of concentration for each zone was estimated by matching the model with the maximum PCE concentration between 1986 and 1991. The calibrated PCE discharge concentrations for all the zones are presented in Table 5.4-1. The agreement between the modeled and observed PCE concentrations is statistically significant as indicated in Table 5.7-2.

The lower correlation coefficients obtained from PCE than for other volatile organic compounds may be due to the decreasing trends observed for PCE at a number of downgradient monitoring wells (see Section 3.6.6.2.6 for a discussion of trend analysis). When calibration is carried out based on the maximum concentrations from 1986 to 1991 and the half life for PCE is assumed to be much higher than the reported values (see Table 5.5-2), this result is expected. However, this indicates that the model is conservatively predicting PCE concentrations as indicated on Table 5.7-2, which shows higher modeled than observed PCE concentrations.

5.7.3.6 Ethyl Benzene

By trial-and-error, the combination of concentration for each zone was estimated by matching the model with the maximum ethyl benzene concentration between 1986 and 1991. The calibrated ethyl benzene discharge concentrations for all the zones are presented in Table 5.4-1. The agreement between the modeled and observed ethyl benzene concentrations is statistically significant except for Case 2 as indicated in Table 5.7-2.

During the course of calibration, it was found that an anomaly exists in the vicinity of wells M36A and M37A where relatively high concentrations of ethyl benzene (approximately 10 µg/L at M36A and 300 µg/L at M37A) have been observed. This was found to be anomalous for the following two reasons: 1) the concentrations of ethyl benzene at two wells upgradient from M36A and M37A (M38A and M39A) are low (non-detect and 1 µg/L, respectively); and 2) ethyl benzene has a higher degree of adsorption than other chemicals such as TCE and PCE (see Tables 5.5-2 and 5.7-1), and yet the front of these two chemicals is found to move more slowly than that of ethyl benzene (based on concentrations at M36A, M37A, M38A, and M39A). For these reasons, data from M36A and M37A were not used in the calibration.

5.7.3.7 Chlordane

By trial-and-error, the combination of concentration for each zone was estimated by matching the model with the maximum chlordane concentration between 1986 and 1991. The calibrated chlordane discharge concentrations for all the zones are presented in Table 5.4-1. The agreement between the modeled and observed chlordane concentrations is statistically significant as indicated in Table 5.7-2.

5.7.3.8 Uranium

The available data for uranium are limited. The available data indicate the presence of background uranium concentration. Since the data for gross alpha radioactivity are also available, these data are used to supplement the uranium data. It is found that a good match is possible using a K_d value as low as 10 L/mg. This value of K_d is considered conservative because Strenge and Peterson (1989) have reported a K_d value for uranium of approximately 50 L/mg for media with more than 30 percent of fine-grained materials (silty and clayey materials).

By trial-and-error, the combination of concentration for each zone was estimated by matching the model with the maximum uranium concentration between 1986 and 1991. The calibrated uranium discharge concentrations for all the zones are summarized in Table 5.4-1. The agreement between the modeled and observed uranium concentrations is statistically significant for Case 1 as indicated in Table 5.7-2. The marginal correlation between the modeled and observed uranium concentrations implies that the assumptions made to model uranium such as the landfill being a continuous source of uranium since the beginning of waste disposal, although conservative for baseline risk assessment purposes, may not be realistic. In other words, the model significantly over-estimates the release of uranium from the landfill and its movement in ground water.

5.7.3.9 Tritium

Data for tritium were obtained from a limited number of wells. Calibration was conducted only to match the available data. By the trial-and-error method, the combination of concentration for each zone was estimated by matching the model with the maximum tritium concentration between 1986 and 1991. The calibrated tritium discharge concentrations for all the zones are presented in Table 5.4-1. As in the case of uranium, the correlation between the modeled and observed tritium concentrations is significant for Case 1 (see Table 5.7-2) but not the other cases. This again implies that the assumptions made in the contaminant transport model may not adequately represent the tritium discharge and transport even though these assumptions are conservative for the purpose of baseline risk assessment.

5.8 PREDICTIVE ANALYSIS

The simulations conducted for the model calibration cover the time period between the commencement of waste disposal (approximately 1957) and 1986, when the subsurface barrier along Hawthorne Boulevard was installed. Predictive analysis simulation starts in 1986 and covers various future periods. The short-term predictive analysis covers the period between 1986 and 1991, and the long-term predictive analysis projects the concentrations up to 400 years in the future for use in the baseline risk assessment. The simulated concentration distributions of each principal chemical in 1986 was used as the initial condition for predictive analysis simulation. Prior to conducting the chemical migration simulation, the Hawthorne Boulevard barrier was incorporated into the flow model so that the simulated flow reflects the existence of the barrier. Simulation of the migration of each principal chemical was conducted to evaluate the chemical concentration at the locations of potential receptors. An additional degree of conservativeness is incorporated into the model by ignoring the extraction wells which have been removing ground water collected upgradient of the barrier since 1986.

5.8.1 Potential Receptors

Five locations were selected as the potential receptors (locations which could have public access or exposure) for predictive analysis with the calibrated ground water contaminant transport model and subsequent baseline risk assessment (Section 6.0). The following criteria were used to select the potential receptors within the model area:

1. Active domestic, industrial, and/or municipal supply ground water wells within the model area were identified as potential receptors. The California Department of Water Resources, Southern Division provides names and locations of reported active wells in the West Coast Basin (CDWR, 1987, 1990, and 1991). Using these references, only two active supply wells were identified within the model area. One of these wells is currently inoperative.

2. Domestic, industrial, and/or municipal supply wells within the study area which are known by other regulatory agencies to be active but which are not identified by

the CDWR as potential receptors. None of these well types were identified within the model area.

3. Locations proposed by local agencies for planned future domestic, industrial, and/or municipal supply wells within the model area were investigated as potential receptors. None of these well types were identified in the model area.

4. Hypothetical well locations nearest the PVLF which could provide an adequate supply of ground water for domestic, industrial, or municipal purposes. Adequate supply is defined as a constant flow rate of at least 100 gallons per minute. Hypothetical wells are used only if there are an insufficient number of wells near the site that meet any of criteria 1, 2, or 3 above. Three hypothetical wells were identified and used as potential receptors in the predictive analysis using the calibrated contaminant transport model.

Ground water monitoring wells and extraction wells are not considered as potential receptors in this evaluation. Similarly, ground water wells historically used for industrial agricultural or domestic supply within the study area, but have been destroyed or abandoned, are not considered as potential receptors in this evaluation. The five ground water receptors selected for the study are described below and their locations are shown on Figure 5.8-1.

Receptor 1 is listed in DWR (1987, 1990, and 1991) as an active well within the study area. However, the well has reportedly not extracted water since at least 1987, and is reported by DWR to only have a total allotment of approximately 7 acre feet remaining for future pumping. According to Mr. John Bauman, owner of the Palos Verdes Begonia Farm, the well was installed in the 1930's for irrigation purposes, but for the last several years, the well has been inoperative due to a broken pump and infilling of sand. He does plan to refurbish the well in the future.

Receptor 2 is a hypothetical well located just north of the Palos Verdes fault zone near Hawthorne Boulevard. This location was selected to represent an area nearest the PVLF and Hawthorne Boulevard, where an adequate supply of water could be extracted in the future. Receptor 3 is a hypothetical well located just north of the Palos Verdes fault zone at Portola Park. This location was selected to represent an area nearest the north-central boundary of the PVLF where an adequate supply of water could be extracted in the future. Receptor 4 is a hypothetical well located just north of the Palos Verdes fault zone near Crenshaw Boulevard. This location was selected to represent an area nearest the PVLF and Crenshaw Boulevard where an adequate supply of water could be extracted in the future. No active wells currently exist in the three areas described above or are planned for these areas by water agencies. Wells drilled on the landfill side of the Palos Verdes fault zone would likely not be capable of providing adequate supplies of ground water because of the lack of a sufficiently thick sequence of water-bearing, transmissive strata.

Receptor 5 is listed in CDWR (1987, 1990, and 1991) as an active well that is located on the PVLF side of the Palos Verdes fault zone within the study area. This well, known as the Chandler well (Chandler Sand and Gravel Quarry), is an active industrial supply well which extracted 214 acre-feet of ground water in 1991 (CDWR, 1991). Water from this well is pumped into a golf course lake for storage, and then used by the Chandler quarry for industrial purposes and for golf course irrigation.

No potential receptor wells are selected in the residential area immediately downgradient of the PVLF which is bounded by Hawthorne and Crenshaw Boulevards, the Palos Verdes fault zone, and the landfill for the following reasons:

1. No reported active domestic or industrial supply wells exist in this area, and there are no stated plans for future wells.

2. The hydrogeology of this area would not support a well with a large enough capacity (at least 100 gpm) to supply multiple users. The ground water is found mostly in the fracture network of the Monterey Formation bedrock, which has low transmissive properties. It is not until several hundred feet east of the landfill, toward the existing Chandler well (Receptor 5), where relatively thick deposits of water-bearing materials occur, and wells can be pumped successfully.

NOTE: Potential receptors 1 and 5 are industrial water supply wells. Potential receptors 2, 3 and 4 are hypothetical wells.

FIGURE 5.8-1

LOCATION OF POTENTIAL GROUND WATER RECEPTORS

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3. The natural quality of the ground water may not be potable because of the naturally high mineral content and total dissolved solids and the naturally occurring hydrocarbons in the Monterey Formation in the ground water.

4. The City of Torrance tap water is readily available to residents and businesses in this area. The tap water originates from outside the model area. There would be no logical reason for any homeowners to attempt to drill their own wells when City water is readily available.

5.8.2 Short-Term Predictive Analysis

The short-term predictive analysis was conducted to compare the model with the available data between 1986 and 1991 to ensure that the model is consistent with the data. This comparison may be regarded as verification of the calibrated contaminant transport model. The results reveal that the movement of nitrate, a very mobile chemical, in the vicinity of the PVLF during the period of 1986 to 1991 is very slow and most of the transport during this period occurs in the top layer (see Exhibits 5.8-1 and 5.8-2). Similar conclusions may be drawn with regard to vinyl chloride (Exhibits 5.8-3 and 5.8-4). The effects due to injection in the two injection wells, GSF-1 and GSF-2, are observed in the second layer. Nevertheless, concentration attenuation with depth is evident. Prediction of chemical migration for the five year (1986 through 1991) period and the simulated concentration distributions at the end of 1991 for all principal chemicals are included in Appendix E.3.

The simulated concentration distribution at the end of 1991 for each principal chemical was compared against the average concentration of the respective chemical between 1986 and 1991. This comparison is considered appropriate because the movement of chemicals in the vicinity of the PVLF during the five year period is relatively slow. Averaging concentrations at each monitoring well for the 1986 through 1991 period tends to smooth out effects due to spatial variability of chemical influx and migration, and tends to provide a better picture of the trends of chemical transport at the site. Statistics relating to the short-term predictive analysis are shown in Table 5.7-2, Cases 2 and 3. Case 2 corresponds to a comparison between the simulated concentration distribution in 1991 and the mean observed concentration from 1986 to 1991 for each principal chemical. Case

3 corresponds to a comparison between the simulated concentration distribution in 1991 and the maximum observed concentration from 1986 to 1991 for each principal chemical.

A comparison between the correlation coefficients in Cases 1 and 3 indicates little change in model correlation within the period of five years (Table 5.7-2). Upon comparing the correlation coefficients of Cases 1 and 2, one finds that the transport model correlates better with the maximum concentration, mainly because the model was calibrated using the maximum concentration.

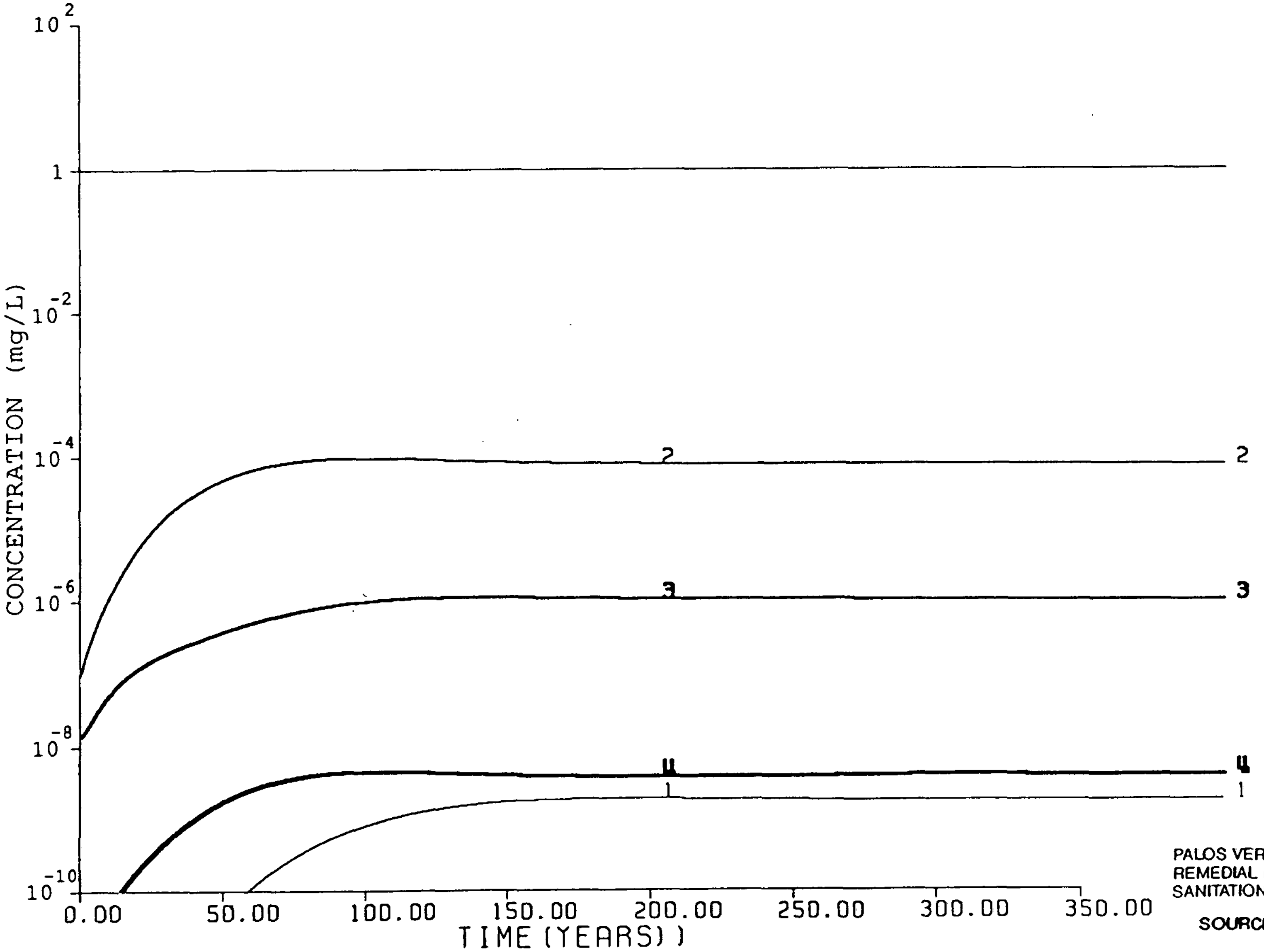
The overall trend of the transport model may be inspected from the variation in average simulated concentrations in 1986 (Case 1) and in 1991 (Case 2). For all the principal chemicals, the trend indicates an increase (except for chlordane, which is relatively immobile) in simulated chemical concentrations between 1986 and 1991. The observed concentrations, however, indicate a decreasing trend for almost all principal chemicals except for chlordane, uranium, and tritium. The discrepancy between the trends for modeled and observed concentrations is primarily due to the conservative assumptions made in the contaminant transport model such as continuous release of chemicals from the landfill since waste disposal began. Although these assumptions may not be realistic, the resulting contaminant transport model tends to overestimate the chemical concentrations and results in conservative (high) risks. The overestimation factor ranges from 1.09 to 69. The overestimation factor in this study is defined as the ratio of mean simulated chemical concentration to mean observed concentration of the same chemical. Because of the conservative assumptions and parameters used in the long-term predictive analysis, the model is expected to be conservative in the long-term predictive mode.

5.8.3 Long-Term Predictive Analysis

The baseline risk assessment described in Section 6.0 was conducted using results from the long-term predictive analysis. The time period of 300 to 400 years was found, through trial and error, to be adequately long to capture peak concentrations at potential receptors for organic chemicals, and adequately long to detect steady-state concentration for inorganic chemicals. Breakthrough curves (concentration versus time curves) of nitrate at the locations of potential receptors in Layers 1, 2, and 3 are shown in Figures 5.8-2 through 5.8-4, respectively. These figures

FIGURE 5.8-2

PREDICTED BREAKTHROUGH CURVES OF NITRATE CONCENTRATION AT POTENTIAL RECEPTORS, LAYER 1



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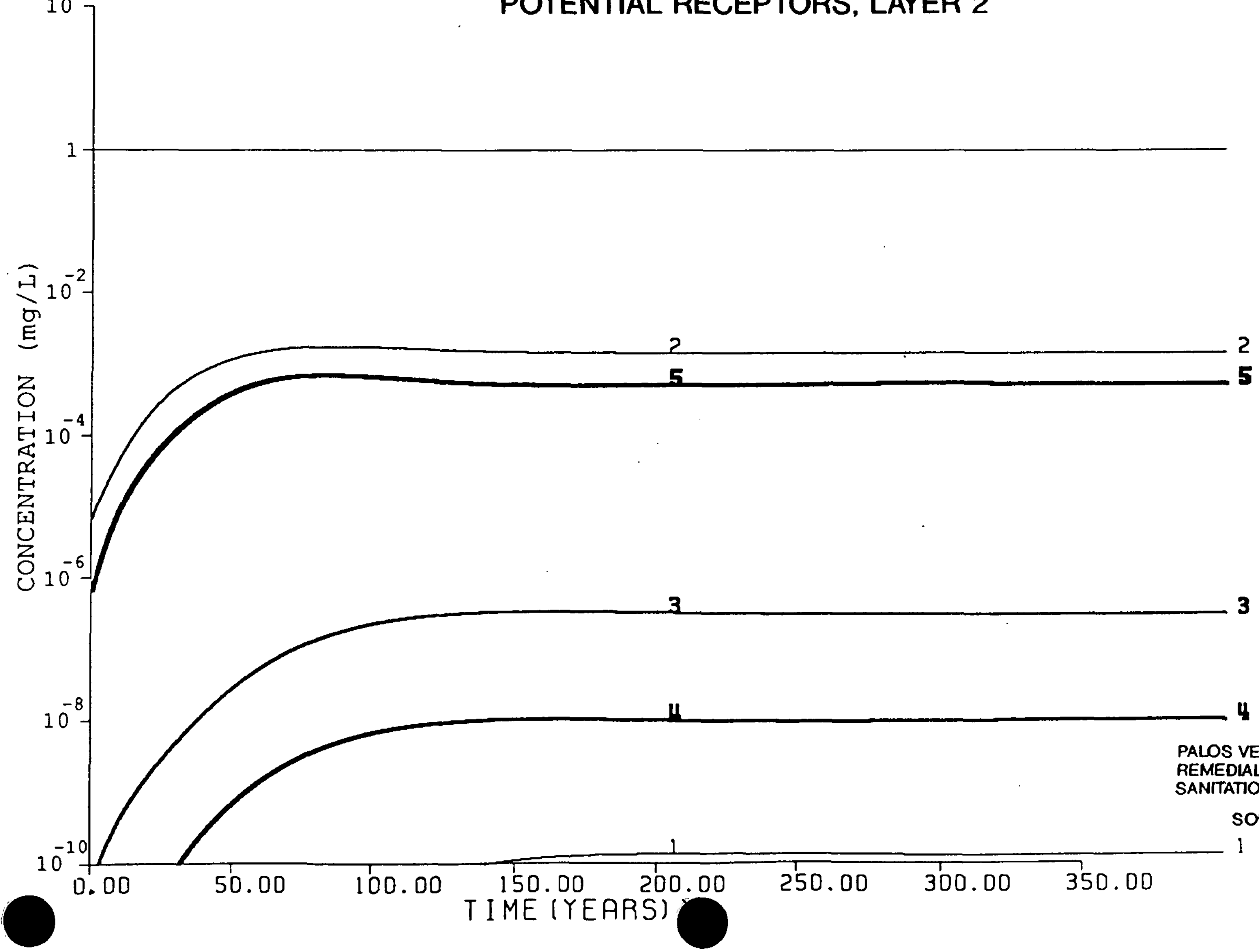
SOURCE: DAMES & MOORE, INC.

FIGURE 5.8-3

PREDICTED BREAKTHROUGH CURVES OF NITRATE CONCENTRATION AT POTENTIAL RECEPTORS, LAYER 2



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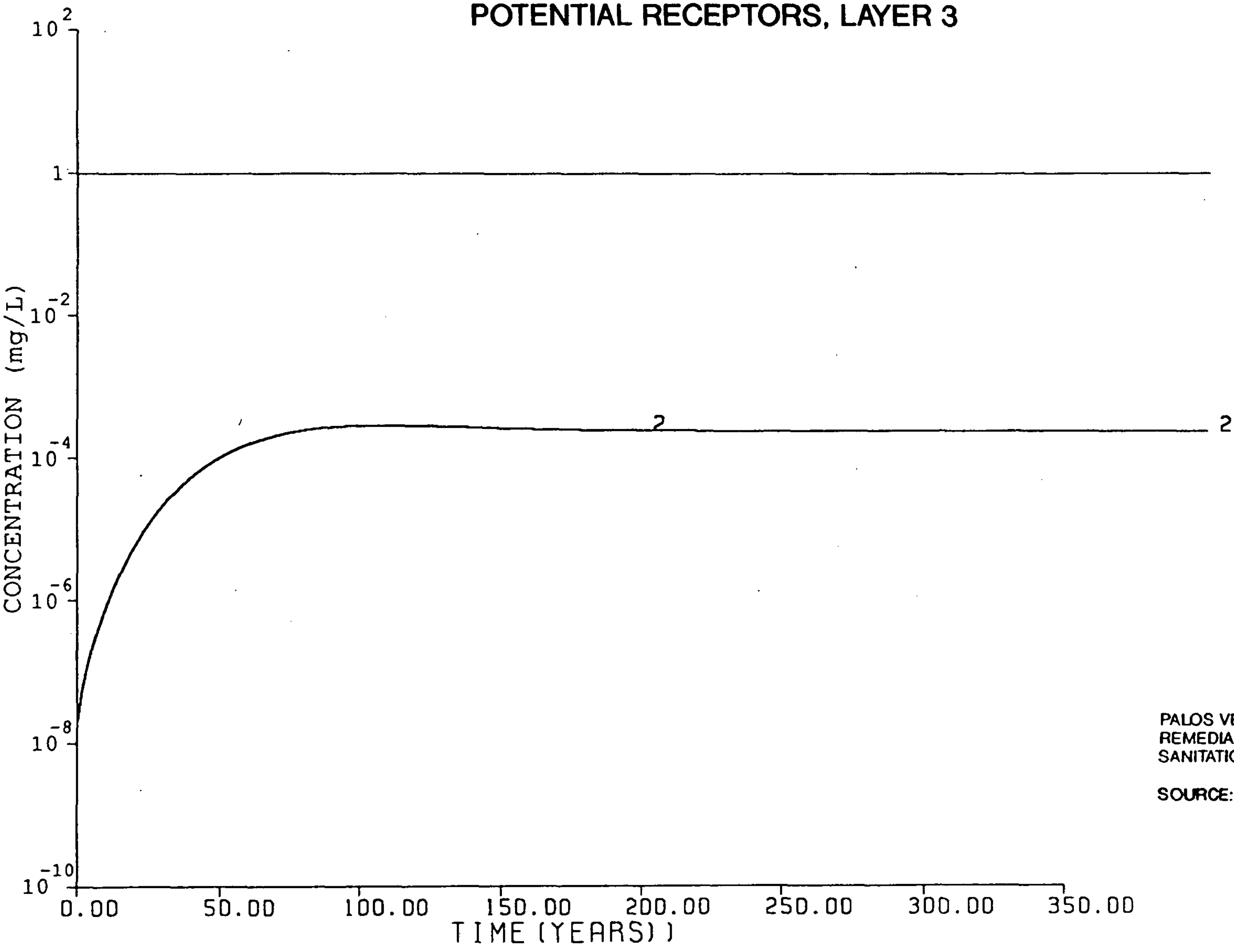
SOURCE: DAMES & MOORE, INC.

FIGURE 5.8-4

PREDICTED BREAKTHROUGH CURVES OF NITRATE CONCENTRATION AT POTENTIAL RECEPTORS, LAYER 3



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indicate that steady-state concentrations occurs approximately 80 years after the long term predictive analysis begins (1986) at some of the receptors and the highest concentration level occurs at Receptor 2 in Layer 2.

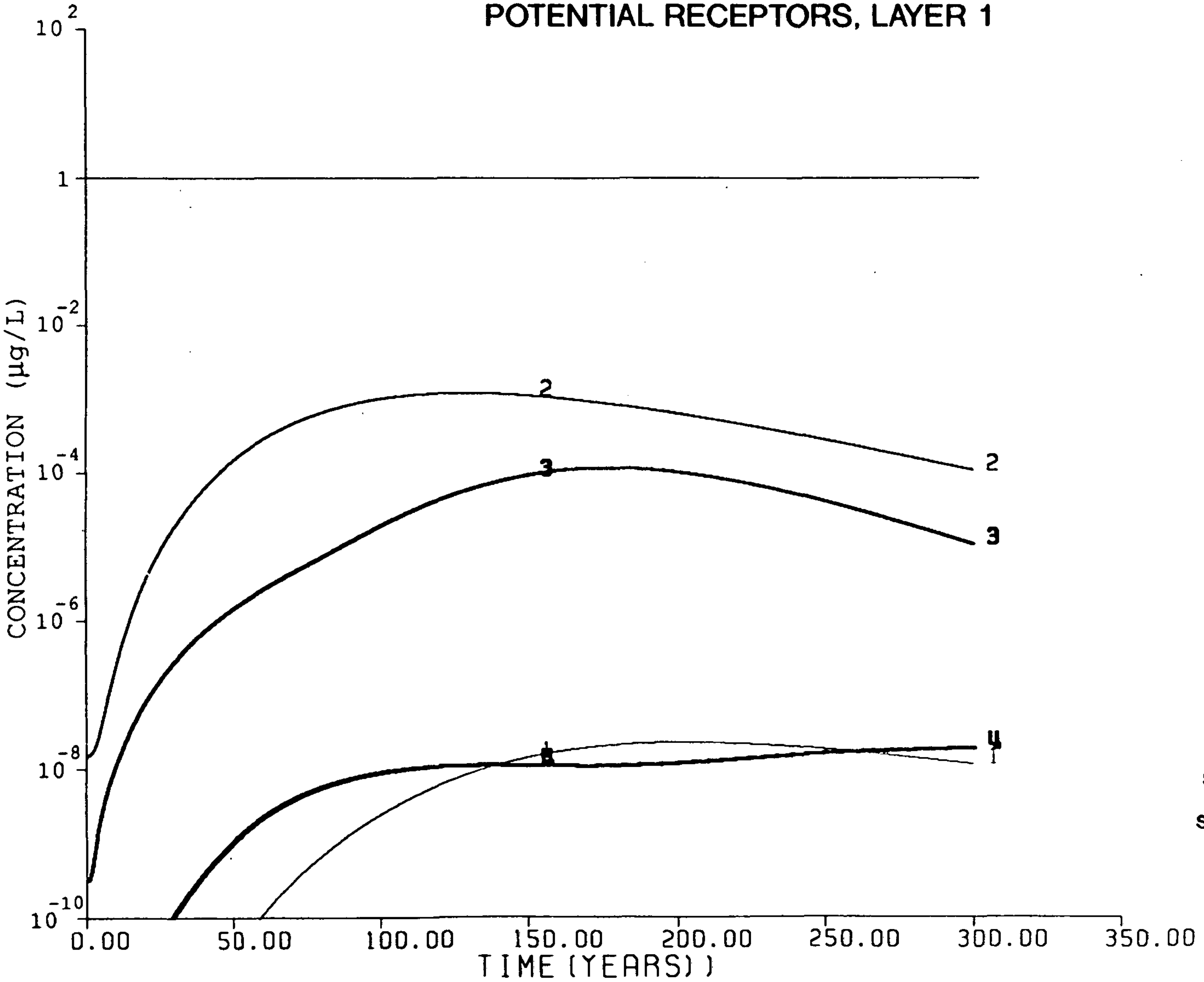
Breakthrough curves of vinyl chloride at the locations of potential receptors in Layers 1, 2, and 3 are shown in Figures 5.8-5 through 5.8-7, respectively. An inspection of these figures reveals that the concentration level is highest at Receptor 2 in Layer 2. The temporal variation of concentration of vinyl chloride is somewhat different from that of nitrate. The concentration of vinyl chloride tends to peak at least 100 years after the simulation begins. This is thought to be attributable to the movement of the plume created by the injection wells (GSF-1 and GSF-2) as well as biodegradation.

Breakthrough curves and concentration distributions for all principal chemicals are presented in Appendix E.3. These distributions represent the approximate maximum extent of migration for all modeled chemicals except uranium. For uranium, because of its long half-life, the maximum extent of migration is still increasing after 300 years. However, because of the low level of predicted uranium concentration at the potential receptors after 300 years, it was deemed adequate to terminate simulation after 300 years. In addition, the results of nitrate transport simulation (which is similar to uranium without adsorption) suggest that the breakthrough concentration of uranium at receptors would also likely be low.

In general, the migratory pattern of chemical plumes emanating from the assumed chemical sources within the PVLF simply follows the direction of the general hydraulic gradient. Once the plumes reach the Palos Verdes fault zone they tend to migrate in the direction parallel to the fault axis. It is possible that small amounts of chemicals may be transported through the Palos Verdes fault zone to the West Coast Basin after tens of years. Very low simulated chemical concentrations (a small fraction of μg/L) are estimated by the model to occur at potential receptors in the West Coast Basin. This observation is consistent with the hydrogeologic hypothesis that the Palos Verdes fault zone functions as a partial hydrogeologic barrier.

FIGURE 5.8-5

PREDICTED BREAKTHROUGH CURVES OF VINYL CHLORIDE CONCENTRATION AT POTENTIAL RECEPTORS, LAYER 1



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SOURCE: DAMES & MOORE, INC.

FIGURE 5.8-6

PREDICTED BREAKTHROUGH CURVES OF VINYL CHLORIDE CONCENTRATION AT POTENTIAL RECEPTORS, LAYER 2

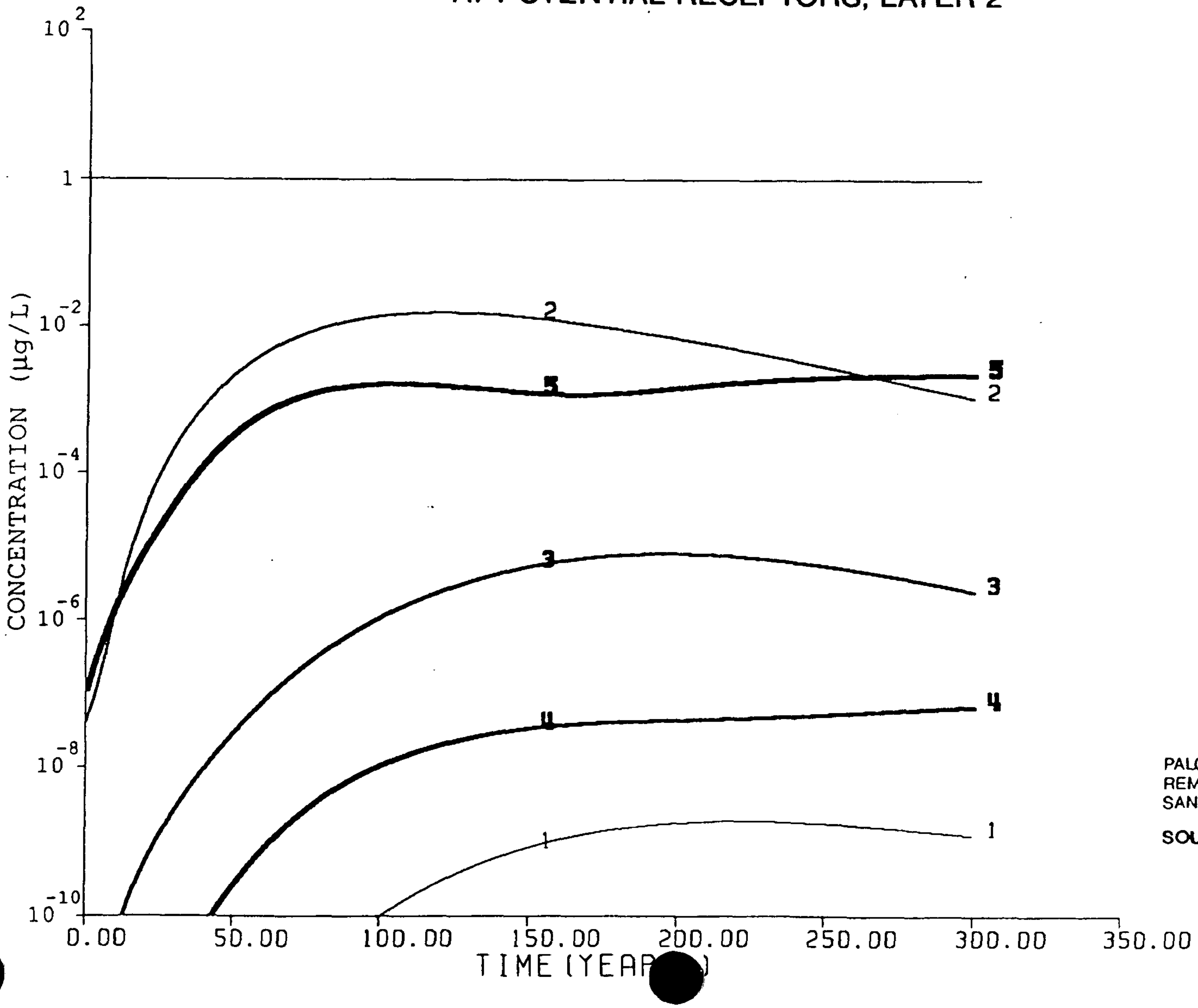

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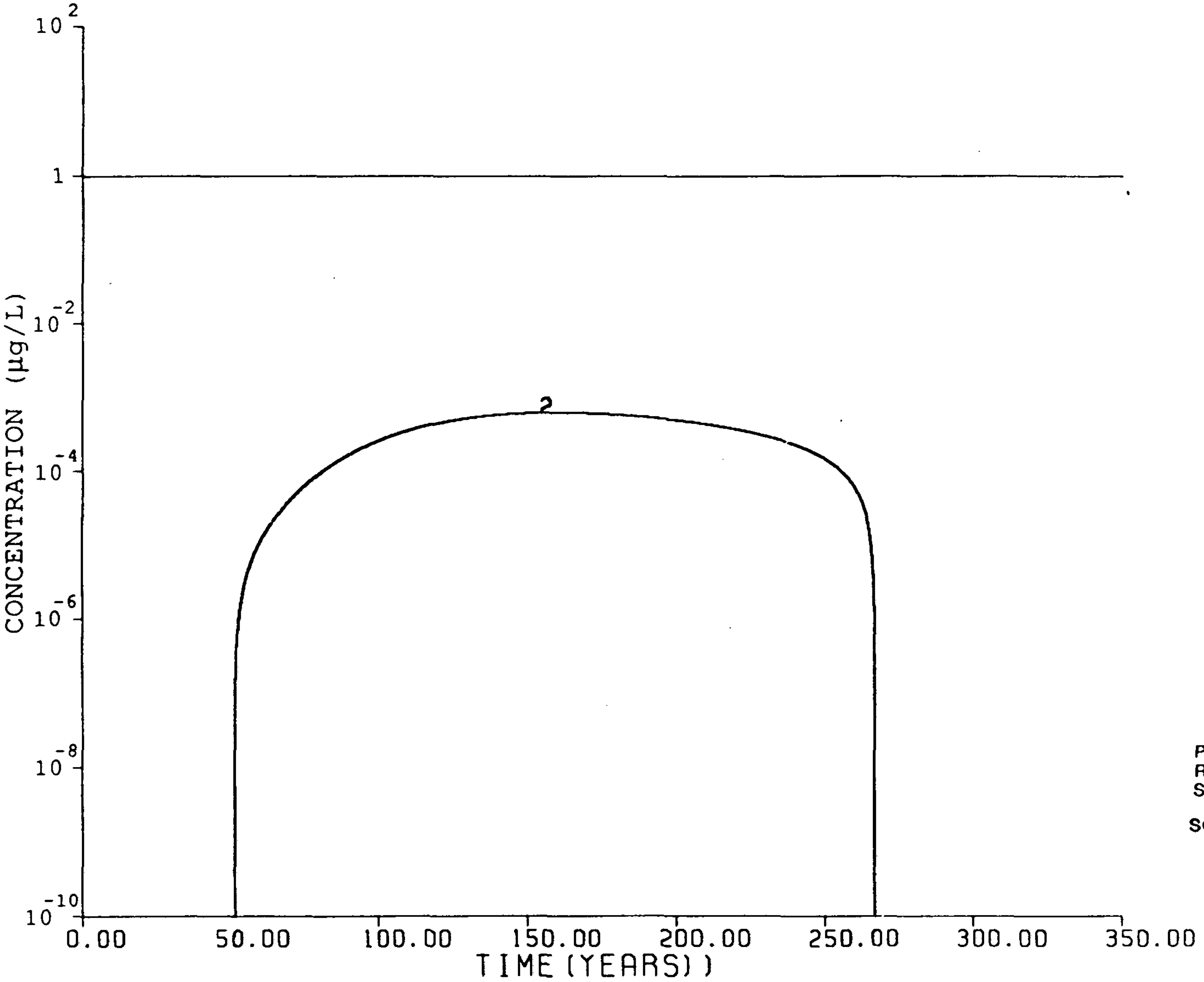
FIGURE 5.8-7

PREDICTED BREAKTHROUGH CURVES OF VINYL CHLORIDE CONCENTRATION AT POTENTIAL RECEPTORS, LAYER 3



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5.9 SENSITIVITY ANALYSIS

Sensitivity analysis was the process of assessing the effects on the model's response due to prescribed changes in hydrogeologic and transport parameters. A sensitivity analysis was conducted to assess the potential effects of uncertainty associated with the parameters of the calibrated model.

There are two major parameter sets that could affect the distribution of chemical concentrations and the level of chemical concentrations at the potential receptors: flow parameters and transport parameters. An extensive sensitivity analysis has been conducted for the flow model (see Appendix E.2). Results indicate that the variation in hydraulic conductivity and recharge do not change the major flow direction in the vicinity of the landfill. Sensitivity analysis cases with modified flow and transport parameters are described below. Vinyl chloride was chosen as the principal chemical for the sensitivity analysis.

Case	Description
0	Base Case - No parameters are modified.
1	Longitudinal dispersivity is increased by a factor of two.
2	Effective porosity for all formations is decreased by 50 percent.
3	Half-life is set at 22 years for vinyl chloride.
4	Half-life is set at eight years for vinyl chloride.
5	Downward vertical gradient is imposed on side boundaries of the model. The following constraints along the boundaries upgradient from the landfill are simultaneously imposed. • Heads in Layer 1 remain unchanged. • Heads in Layer 2 equal heads in Layer 1 minus 40 feet. • Heads in Layer 3 equal heads in Layer 2 minus 40 feet. • Heads in Layers 4 and 5 equal heads in Layer 3.
6	Hydraulic conductivities of the Overburden (Qo) and the Undifferentiated Sand of the San Pedro Formation (Qus) are increased by a factor of two.
7	Hydraulic conductivities of the Malaga Mudstone (Tmm) and the Valmonte Diatomite (Tmv) are increased by a factor of two.

8 Random errors are introduced in the constant head boundaries of the flow model. Heads along side boundaries are assumed to have a random noise (error) which is assumed to be uniformly distributed and vary between +25 feet to -25 feet upgradient from the fault and +2.5 to -2.5 feet downgradient from the fault (i.e., the error has an equal probability of occurrence anywhere between -25 feet and +25 feet, or +2.5 to -2.5 feet). The ranges of 50 and 5 feet are approximately equal to 20 percent of the ranges of observed ground water elevations upgradient and downgradient of Palos Verdes fault zone, respectively.

9 The Palos Verdes fault zone in Layers 1 and 2 is partially removed, thereby introducing localized fault leakage.

10 Calibrated f_{oc} values in Section 5.7 are used, which represent a decrease by a factor of two to three from the original f_{oc} values.

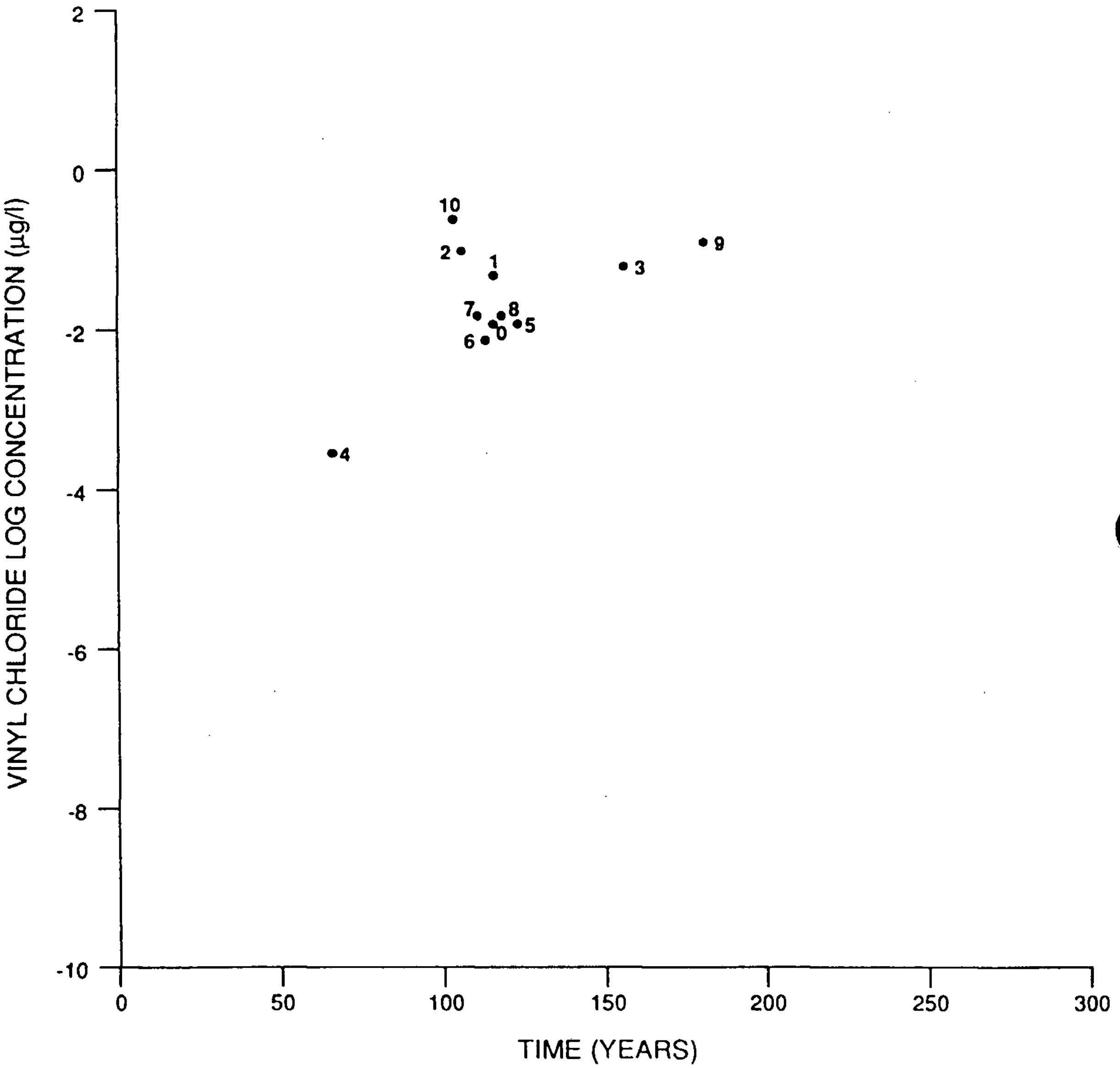
Cases 1, 2, 3, 4, and 10 are associated with transport parameters. All cases, except for Case 4, are expected to increase the concentration levels at the potential receptors. For example, an increase in the value of dispersivity or a decrease in effective porosity or a decrease in the values of f_{oc} would cause the chemical front to reach the receptors sooner. Cases 5 to 9 are associated with the flow model and flow parameters. Cases 6 and 7 are expected to have the same effects as Cases 1, 2, and 10.

Breakthrough curves of the above sensitivity analysis cases are presented in Appendix E.3. The results indicate that Receptors 2 and 5 in Layer 2 are the most critical. The variation of concentration at the above receptors is diagrammatically summarized in Figures 5.9-1 and 5.9-2, which show the variation of vinyl chloride concentration at Receptors 2 and 5, respectively. As shown in these figures, the range of concentration uncertainty spreads over three orders of magnitude, or about 1.5 orders of magnitude about the Base Case. Cases 9 (fault removal) and 10 (f_{oc} varied) appear to be the most critical.

The effects due to uncertainty in the distribution of ethyl benzene concentration were also investigated. An alternative ethyl benzene distribution was used to replace the distribution in the Base Case. The concentration distribution in the Base Case cannot mimic the anomalously high

FIGURE 5.9-1

SCATTER DIAGRAM OF MAXIMUM CONCENTRATIONS OF VINYL CHLORIDE FROM SENSITIVITY ANALYSIS CASES AT RECEPTOR 2, LAYER 2



LEGEND:

10 ● Maximum Concentration Point. Sensitivity Case # Shown.

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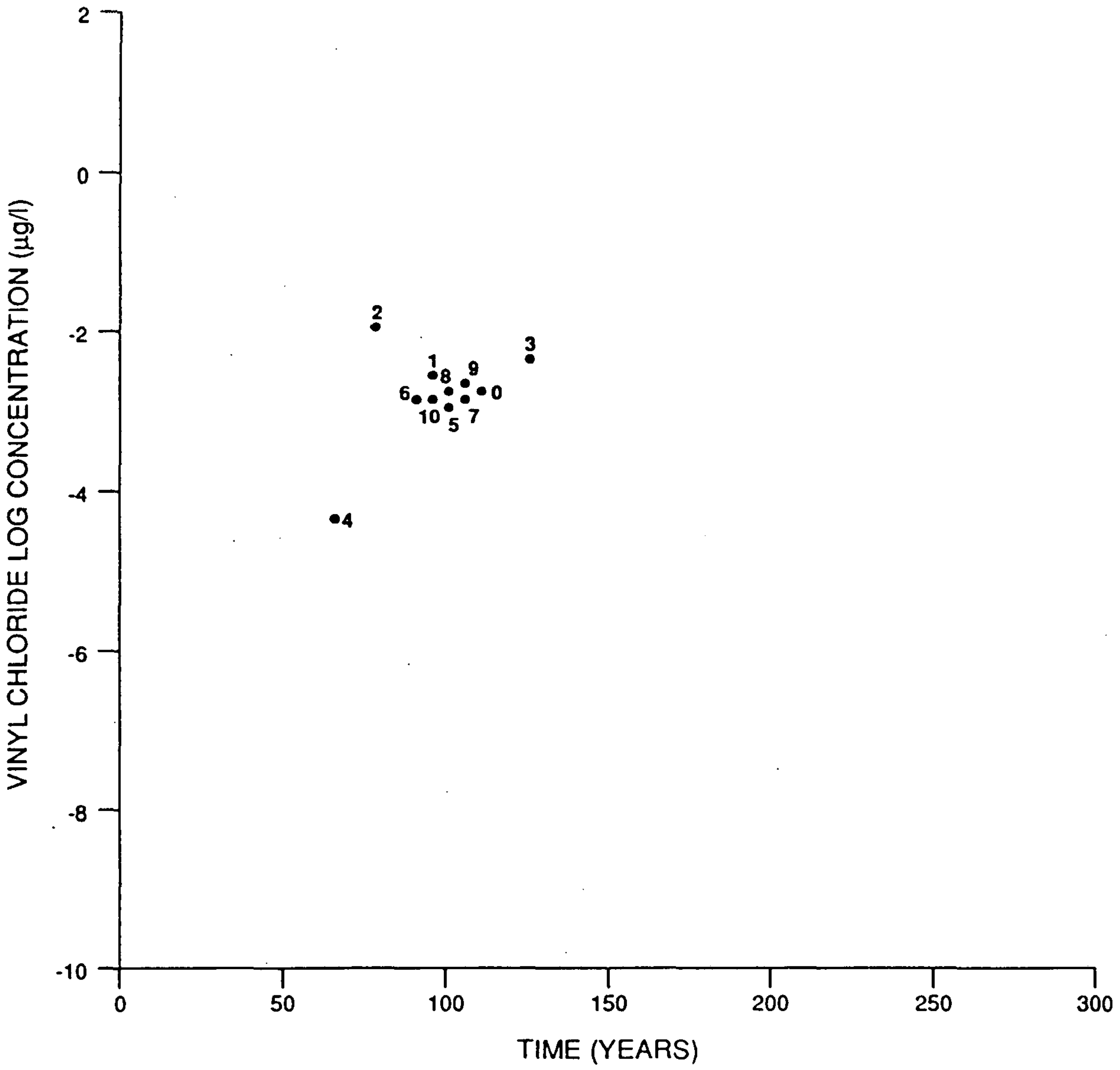
FIGURE 5.9-2

SCATTER DIAGRAM OF MAXIMUM CONCENTRATIONS OF VINYL CHLORIDE FROM SENSITIVITY ANALYSIS CASES AT RECEPTOR 5, LAYER 2



LEGEND:

10● Maximum Concentration Point. Sensitivity Case # Shown.

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concentration of ethyl benzene at M36A and M37A. Concentration in the easternmost parcel was elevated to compensate for the uncertainty outside the PVLF site. A comparison between breakthrough curves of the Base Case and those of the alternative case indicates that the ethyl benzene concentration level at the potential receptors remains equally low at approximately 10^{-6} μg/L. Therefore, the uncertainty associated with the distribution of ethyl benzene does not appear to have a significant impact on the concentration of ethyl benzene at the potential receptors.

5.10 SUMMARY

The ground water contaminant transport modeling exercise is summarized below:

1. The chemical data were found to be spatially and temporally variable. Because of the high degree of data variability, the transport model was developed to simulate the major characteristics of the transport processes at the site.

2. For some chemicals, such as nitrate, chloride, and some metals, variable background concentrations have been detected in upgradient wells. None of the upgradient wells are included in the model calibration.

3. A high degree of uncertainty associated with the metal concentration data was encountered. The cause of uncertainty is primarily due to the presence of suspended solids. Because of the high degree of uncertainty associated with the metal concentration data, and their strong tendency to migrate very slowly in the ground water (due to their high adsorptive properties), metals are not modeled in this study. Two metals, boron and silver, were identified as being relatively mobile, and are similar to nitrate in terms of transport properties. The concentrations for these metals at potential receptors may be scaled from those of nitrate.

4. The flow model was refined using information relating to chemical transport within the study area. The resulting flow model is essentially the same as the previously reported flow model with minor parameter adjustments.

5. Results from a preliminary model verification, based on chemical data between 1986 and 1991, indicate that the model is conservative and has a tendency to overestimate the chemical concentration within the PVLF and its vicinity.

6. The Palos Verdes fault zone, which separates the PVLF area from the West Coast Basin, functions as a partial barrier allowing only relatively small lateral inflow from the Palos Verdes Hills to the West Coast Basin. Water-borne chemicals may locally migrate through the fault zone in areas of relatively high permeability fault gouge. The potential existence of the localized leakage areas is regarded as one of the uncertainties requiring analysis. The sensitivity analysis results suggest that a localized leakage in the fault directly between the receptor and the source could cause the level of chemical concentration to increase by an order of magnitude at the potential receptors.

7. Results from the sensitivity analysis, performed mainly using the vinyl chloride data, suggest that the range of uncertainty of concentration at potential receptors is approximately 1.5 orders of magnitude about the base case (case with best-estimate parameters). The model was found to be sensitive to the variability of adsorptive properties, fault hydraulic properties, and effective porosity.

8. The long term predictive analysis estimates that chemical concentrations at potential receptors will vary between a few tenths of a μg/L to several orders of magnitude lower for organic chemicals; between a few μg/L to several orders of magnitude lower for nitrate; and between a few thousandths of a pCi/L to several orders of magnitude lower for tritium and uranium. A summary of maximum concentrations at potential receptors is presented in Table 5.10-1.

5.11 POST AUDIT

The final step in hydrogeologic modeling is a post audit, which compares model predictions and construction parameters against new field data or field observed conditions some time in the future. The purpose of the past audit is to answer the question "Did the model accurately

TABLE 5.10-1

MAXIMUM CONCENTRATION AT POTENTIAL RECEPTORS

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Chemical	Maximum Concentration	Layer/Receptor
Nitrate	1.6×10^{-3} mg/l	2/2
Vinyl Chloride	2.5×10^{-1} μg/l	2/2
Benzene	1.0×10^{-3} μg/l	2/5
Trichloroethylene	1.2×10^{-2} μg/l	2/5
Tetrachloroethylene	2.2×10^{-3} μg/l	2/5
Ethyl Benzene	3.2×10^{-6} μg/l	2/5
Chlordane	$<10^{-10}$ μg/l	All/All
Uranium	2.5×10^{-6} pCi/l	1/2
Tritium	4.0×10^{-4} pCi/l	2/5

Note: Maximum concentrations and their locations of occurrence are obtained from the breakthrough curves in Appendix E.3.

predict future conditions?" If the answer is "yes," then the modeling effort was a success and the model has been validated for the site. Anderson and Woessner (1992) state that a post audit of short term (*i.e.,* several years) can be useful, but they do not allow sufficient time for the model to move far from the calibrated solution. They suggest that a post audit be performed on model predictions at least ten years after model construction prediction to provide a rigorous test of the model's accuracy.

As new field data are generated, they should be compared to the existing model to ensure that they are generally consistent with the conceptual model. Errors in the conceptual model have led to inaccurate predictions by chemical transport models in the past (Anderson and Woessner, 1992). These errors generally occur because the hydrogeologic system is not completely understood at the time of model construction due to the impracticality of obtaining an exhaustive data set. Assumptions and interpretations, therefore, are made by qualified engineers, geologists, and hydrogeologists to complete the conceptual model based on available data, and this conceptual model is used as the basis for the numerical model.

The uncertainties associated with the numerical models are typically evaluated using uncertainty (sensitivity) analyses. For the original PVLF flow model, 28 separate sensitivity analyses were designed and run. For the revised flow model coupled with the chemical transport model, an additional ten sensitivity analyses were designed and run. Table 5.11-1 lists all the sensitivity runs and the modified parameters for each run. Section 3.4.5.5 presents a full discussion of the sensitivity analysis results for the original flow model. Section 5.9 presents a full discussion of the sensitivity analysis results for the revised flow model and chemical transport model.

As previously discussed, the flow and chemical transport models developed for the PVLF had one main purpose; to provide predictions of conservative (maximum) concentrations of key landfill derived chemicals in ground water at potential receptors for use in the baseline health risk assessment. To meet this purpose, the model was intentionally designed to be conservative wherever possible to provide a worst-case concentration at the receptors. For example, the maximum concentration of each chemical at each well over a five-year period was used for calibration. Conservative assumptions were made during model construction when actual field data were unavailable. For example, predominant horizontal flow was imposed on ground water movement,

TABLE 5.11-1

FLOW AND CHEMICAL TRANSPORT MODEL SENSITIVITY ANALYSES

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MODEL	SENSITIVITY ANALYSIS CASE NO.	MODIFIED PARAMETER
Flow	0	None. This is the Calibrated Model (Base Case).
	1	Increase K value of fill & colluvium by a factor of 10
	2	Increase K value of alluvium by a factor of 10
	3	Increase K value of Qus by a factor of 10
	4	Increase K value of Tmm by a factor of 10
	5	Increase K value of Tmv by a factor of 10
	6	Increase K value of Tma by a factor of 10
	7	Increase K value of Jc by a factor of 10
	8	Created 5 "holes" in the Palos Verdes fault zone
	9	Decrease global recharge by 75 percent
	10	Increase global recharge by 75 percent
	11	Fixed heads at 749A & 240A to simulate pumping
	12	Same as Case 11, but remove pumping well 271N
	13	Minimum K value of 8 x 10^{-6} adjacent of fault
	14	Remove recharge near M59B
	15	Remove pumping from Chandler well 271N
	16	Remove the Palos Verdes fault as a barrier
	17	Increase K value in the PV Hills by a factor of 10
	18	Increase K value in West Coast Basin by a factor of 10
	19	Increase recharge in Zone 1 by a factor of 2
	20	Increase recharge in Zone 2 by a factor of 2
	21	Increase recharge in Zone 3 by a factor of 2
	22	Increase recharge in Zone 4 by a factor of 2
	23	Increase recharge in Zone 5 by a factor of 2
	24	Increase recharge in Zone 6 by a factor of 2
	25	Increase recharge in Zone 7 by a factor of 2

TABLE 5.11-1 (CONTINUED)

FLOW AND CHEMICAL TRANSPORT MODEL SENSITIVITY ANALYSES

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MODEL	SENSITIVITY ANALYSIS CASE NO.	MODIFIED PARAMETER
Flow	26	Increase recharge in Zone 8 by a factor of 2
	27	Increase recharge in Zone 9 by a factor of 2
	28	Increase recharge in Zone 0 by a factor of 2

MODEL	SENSITIVITY ANALYSIS CASE NO	MODIFIED PARAMETER
Chemical Transport	0	None. This is the Calibrated Model (Base Case).
	1	Increase longitudinal dispersivity by a factor of 2
	2	Decrease effective porosity of formations by 50 percent
	3	Half-life for vinyl chloride was set at 22 years
	4	Half-life for vinyl chloride was set at 8 years
	5	Downward vertical gradients were created
	6	Increase K value of Qo and Qus by a factor of 2
	7	Increase K value of Tmm and Tmv by a factor of 2
	8	Random errors in the constant heads were introduced
	9	Partially remove portions of the Palos Verdes fault
	10	Decrease Koc values by a factor of 2 to 3

NOTE: For a complete discussion of the sensitivity analysis results, including summary tables, figures, and maps, refer to sections 3.4.5.5 (flow model) and 5.9 (revised flow model and chemical transport model).

which results in the chemicals reaching the potential receptors sooner than if there is actually a significant vertical component of flow to the system.

Because of its intentionally conservative design, the model is not intended to precisely simulate the conditions at PVLF at any given period of time, nor should it be constantly updated and recalibrated to match current conditions as long as field measured parameters generally fit within the bounds of the construction parameters and sensitivity analyses. Whenever substantial new field data are generated, such as new borings, wells, aquifer tests, water levels, or water quality data, they should be compared against the model input parameters and sensitivity analyses. However, as previously mentioned, complete model reconstruction and recalibration is not warranted until sufficient time has passed for a post audit to be meaningful. An exception is when the new field data do not fit into the model at all, and cause a rethinking and redesign to the conceptual model.

A substantial amount of field data were collected during the additional downgradient hydrogeologic field program after completion of the PVLF models. These data were evaluated against the PVLF flow and transport model parameters and compared to the conceptual model of the site to assess whether the new parameters fit the model, or whether significant differences existed which warranted model recalibration. The remainder of this section will describe the results of this evaluation.

5.11.1 Evaluation of Hydrogeologic Modeling

The original flow model developed in 1992 (described in Section 3.4.5) was constructed based on the hydrogeologic data available at that time. Subsequently, the flow model was used as a basis for the development of a chemical transport model, at which time the ground water flow model was further calibrated to account for both the hydrogeologic data and water quality data. During the second stage of model development, there was no change in the conceptual model; however, the hydraulic properties in the flow model were further adjusted to achieve satisfactory simultaneous correspondence between the two models (flow and chemical transport models) and field observations (ground water elevations and water quality data). The second flow model is believed to more accurately represent real world conditions than the first model. The previous discussion throughout Section 5.0 presented a detailed discussion of the development of the chemical transport model, including the modifications made to of the original flow model. The discussion in the remainder of this section is based on the second flow model and chemical transport model, and how the data generated during the additional downgradient hydrogeologic field program compared to the data used in the models.

As previously stated, the models were developed for the purposes of evaluating conservative chemical concentrations at potential receptors for the PVLF baseline health risk assessment. Uncertainties associated with the data were circumvented or compensated for by a number of conservative assumptions described in detail in Section 6.0.

The additional downgradient hydrogeologic field program was performed after completion of the flow and transport models. The data generated during this additional program was evaluated against data used to construct the models to assess whether the data fit the model parameters, or whether the model needed to be redeveloped and recalibrated using the new data. In order to evaluate the impact of the new data on the developed models, the following issues must be addressed.

a. Conceptual Model

Are the new data consistent with the existing conceptual model?

If the new data are inconsistent with the existing conceptual model, what effects will the inconsistencies have on model predictions?

b. New Parameter Values and Additional Observations

Do the new data agree favorably with the developed model?

c. Uncertainty Scenarios

Do the new data introduce any new uncertainty scenarios?

If so, can the existing modeling results be used to address these new scenarios?

These issues are discussed individually in the following sections.

5.11.2 Conceptual Model

The conceptual model based on the available data prior to 1992 may be described as follows:

- In the PVLF area, the flow in the shallow zone essentially follows the topographical gradient direction towards the Palos Verdes fault zone.
- Owing to the lack of data in the deep zone, the deep-zone flow is currently not precisely known. However, because of the presence of topographical highs in the areas upgradient from the PVLF, the deep-zone flow was thought to be topographically driven and therefore would be similar to the shallow-zone flow along the horizontal plane.
- Because of the apparently steep hydraulic gradient across the Palos Verdes fault zone, the fault is thought to act as a partial hydrogeologic barrier, allowing ground water to discharge very slowly into the West Coast Basin.

Additional ground water elevation data, hydraulic properties data, and water quality data from the new monitoring wells (M63B (AB2) through M70B (AB7)), are not inconsistent with the above features of the conceptual model. However, ground water elevation data from M64B (AB1a) may introduce local uncertainties relating to the local hydraulic functions. The anomalously low ground water elevation in M64B (AB1a) suggests the following hypotheses:

Hypothesis 1 There may be a channel or trough traversing this well and acting as a drain to the local flow system. If this channel does exist, it is likely to drain the local ground water in a direction subparallel to the fault axis toward the fault. This particular direction was deduced from the fact that ground water elevations in wells M63B (AB2) and M51B (RFB4) are higher than that observed in M64B (AB1a), and that the San Pedro Sand is locally present in the area between the main site of the PVLF and the fault zone.

This hypothesis does not contradict the current conceptual model. Although the local channelization in the vicinity of M64B (AB1a) has not been explicitly described the current model, it was addressed in the modeling report as part of data uncertainty associated with the Qo (overburden alluvium) and Qus flow zones. There are no data to support nor refute this hypothesis.

The uncertainty arising from this hypothesis is discussed in Section 5.11.4.

Hypothesis 2 The bedrock (Tmm [Malaga Mudstone] and/or Tmv [Valmonte Diatomite]) beneath M64B (AB1a) is acting as a sink. In other words, immediately below the Qus in the vicinity of M64B (AB1a) is a highly fractured zone of extremely high conductivity and hydraulically connected to an area of low ground water elevation.

Hypothesis 2 is unlikely because the hydraulic conductivity values in these two zones have been found to range from 7 x 10^{-5} to 7 ft/day. For the sink area to be able to absorb a large volume of converging ground water flow from a large area, the hydraulic conductivity of the sink area must be extremely large (hundreds of ft/day or greater).

5.11.3 New Parameter Values and Additional Observations

Additional hydrogeologic data were developed during the additional remedial investigations. This section discusses how the ground water elevations, hydraulic conductivity values, effective porosity values, and water quality data compare to the previous hydraulic data used to model the site.

Ground Water Elevations

Most of the new ground water elevation data were found to be consistent with the model. Although ground water elevations may change over time, the driving force for ground water flow is the hydraulic gradient. A comparison of Exhibit 3.4-3 (March/April 1991 ground water elevation contour map used for model calibration) and Exhibit 3.4-4 (ground water elevation contour map based on the new data) show similar gradients. Thus, ground water flow directions and gradients simulated by the model should continue to adequately simulate real world conditions without modifications. The apparently anomalous data at M64B (AB1a) was addressed in Section 5.11.2.

Hydraulic Conductivity Values

The hydraulic conductivity results obtained during the January 1994 additional downgradient hydrogeologic field program were compared to results obtained during the upgradient and downgradient hydrogeologic field program of the RI and previous hydrogeologic investigations to further identify the hydrogeologic properties of local flow zones, and to assess whether the newer values matched reasonably well with parameters established in the ground water flow model developed for the site. The comparison between the new hydraulic conductivity values and the model calibrated hydraulic conductivity values suggests that, in general and within an order of magnitude, the calibrated and newly-acquired hydraulic conductivity values agree favorably.

The hydraulic conductivity test data from the previous investigations, the downgradient and upgradient hydrogeologic field program of the RI, and the additional downgradient hydrogeologic field program are summarized on Table 5.11-2. Table 5.11-3 presents a comparison of these data to the hydraulic conductivity results derived from the Cooper, Bredehoeft, and Papadopolous (CBP) method for the slug tests conducted during the additional downgradient hydrogeologic rield program. Since the additional data were obtained from wells screened across the Qus, Tmm, Qo/Tmm, and Qus/Tmm flow zones, only the previous results obtained from these same formations are presented in Table 5.11-3.

The hydraulic conductivity results from the additional downgradient hydrogeologic field program for the Qus flow zone (M67B (AB8) and M68B (AB9)) are over a factor of three higher than the previous highest value for this unit. However, it is expected that the true aquifer thickness at the locations of these two wells is greater than that used in the hydraulic conductivity-transmissivity calculation, which would make the value smaller, possibly falling within the range shown in Table 5.11-3 . For example, if the aquifer thickness is 100 feet instead of sixteen feet, which was the saturated screened interval, hydraulic conductivity values at M67B and M68B would be about 3 ft/day

TABLE 5.11-2 (Page 1 of 4)

HYDRAULIC CONDUCTIVITY DATA FOR THE PVLF AREA

PALOS VERDES LANDFILL - REMEDIAL INVESTIGATION REPORT

BORING OR WELL	K (cm/sec)	K (ft/day)	TEST TYPE	GEOLOGIC UNIT	DATA SOURCE
P-1	1.00E-05	2.83E-02	Permeameter	PVLF Cover	Woodward-Clyde, 1981
P-2	3.40E-05	9.64E-02	Permeameter	PVLF Cover	Woodward-Clyde, 1981
P-3	1.00E-05	2.83E-02	Permeameter	PVLF Cover	Woodward-Clyde, 1981
C-5	4.00E-08	1.13E-04	Remold	Qo	Stone, 1975
C-5	3.00E-07	8.50E-04	Sieve	Qo	Stone, 1975
LE-1	7.20E-04	2.04E+00	Aquifer	Qo	Sanitation Districts, 1986a
M37A	1.24E-04	3.51E-01	Slug	Qo	Kleinfelder, 1988
M48A	3.70E-05	1.05E-01	Slug	Qo	Kleinfelder, 1988
M55B (RFB21)	1.62E-05	4.59E-02	Lab	Qo	Herzog, 1991b
RFB22	1.08E-07	3.06E-04	Lab	Qo	Herzog, 1991b
M61B(RFB31)	1.40E-05	3.97E-02	Lab	Qo	Herzog, 1991b
M61B (RFB31)	2.00E-06	5.67E-03	Lab	Qo	Herzog, 1991b
RFB32	1.83E-08	5.19E-05	Lab	Qo	Herzog, 1991b
M36A	1.20E-03	3.40E+00	Slug	Qo/Tmm	Kleinfelder, 1988
M38A	8.50E-05	2.41E-01	Slug	Qo/Tmm	Kleinfelder, 1988
M53B (RFB16)	1.40E-05	3.97E-02	Aquifer	Qo/Tmm	Herzog, 1991a
M69B (AB6)	3.50E-03	9.91E+00	Slug	Qo/Tmm	Dames & Moore, 1994
M70B (AB7)	2.19E-04	6.20E-01	Slug	Qo/Tmm	Dames & Moore, 1994
M23A	5.50E-05	1.56E-01	Slug	Qo/Tmv	Kleinfelder, 1988
M25A	4.20E-05	1.19E-01	Slug	Qo/Tmv	Kleinfelder, 1988
M41A	3.34E-04	9.47E-01	Slug	Qo/Tmv	Kleinfelder, 1988
M44A	3.55E-03	1.01E+01	Slug	Qo/Tmv	Kleinfelder, 1988
M46A	3.80E-05	1.08E-01	Slug	Qo/Tmv	Kleinfelder, 1988
M49A	1.30E-05	3.69E-02	Slug	Qo/Tmv	Kleinfelder, 1988
A-5	1.60E-05	4.54E-02	Permeameter	Qus	Stone, 1975
A-8	3.60E-06	1.02E-02	Permeameter	Qus	Stone, 1975
C-1	6.17E-06	1.75E-02	Remold	Qus	Stone, 1975
C-1	9.00E-06	2.55E-02	Sieve	Qus	Stone, 1975
C-9	2.10E-03	5.95E+00	Remold	Qus	Stone, 1975
M26A	9.90E-06	2.81E-02	Slug	Qus	Kleinfelder, 1988
M52B (RFB13)	6.60E-05	1.87E-01	Aquifer	Qus	Herzog, 1991a
M52B (RFB13)	6.62E-04	1.88E+00	Lab	Qus	Herzog, 1991a
RFB14	3.60E-04	1.02E+00	Lab	Qus	Herzog, 1991a
RFB17	8.06E-04	2.28E+00	Lab	Qus	Herzog, 1991a
M50B (RFB3)	1.75E-03	4.96E+00	Aquifer	Qus	Herzog, 1991a
M50B (RFB3)	9.10E-04	2.58E+00	Lab	Qus	Herzog, 1991a
M51B (RFB4)	1.20E-04	3.40E-01	Aquifer	Qus	Herzog, 1991a
M67B (AB8)	6.69E-03	1.90E+01	Slug	Qus	Dames & Moore, 1994
M68B (AB9)	6.98E-03	1.98E+01	Slug	Qus	Dames & Moore, 1994
M63B (AB2)	1.31E-04	3.70E-01	Slug	Qus/Tmm	Dames & Moore, 1994
M64B (AB1a)	1.81E-03	5.14E+00	Slug	Qus/Tmm	Dames & Moore, 1994
BC-2	3.00E-06	8.50E-03	Packer	Bedrock	Sanitation Districts, 1986a
BC-2	5.00E-06	1.42E-02	Packer	Bedrock	Sanitation Districts, 1986a
BC-2	2.00E-06	5.67E-03	Packer	Bedrock	Sanitation Districts, 1986a
BC-3	3.00E-06	8.50E-03	Packer	Bedrock	Sanitation Districts, 1986a
BC-3	5.00E-06	1.42E-02	Packer	Bedrock	Sanitation Districts, 1986a
BC-3	4.00E-06	1.13E-02	Packer	Bedrock	Sanitation Districts, 1986a
C-3	2.00E-06	5.67E-03	Sieve	Bedrock	Stone, 1975
Parcel 6	5.00E-07	1.42E-03	Field Perc.	Bedrock	Stone, 1975

TABLE 5.11-2 (CONTINUED - Page 2 of 4)

HYDRAULIC CONDUCTIVITY DATA FOR THE PVLF AREA

PALOS VERDES LANDFILL - REMEDIAL INVESTIGATION REPORT

BORING OR WELL	K (cm/sec)	K (ft/day)	TEST TYPE	GEOLOGIC UNIT	DATA SOURCE
A-2	1.60E-06	4.54E-03	Permeameter	Tmm	Stone, 1975
A-3	2.10E-06	5.95E-03	Permeameter	Tmm	Stone, 1975
A-4	2.70E-07	7.65E-04	Permeameter	Tmm	Stone, 1975
A-9	1.30E-05	3.69E-02	Permeameter	Tmm	Stone, 1975
C-1	6.10E-08	1.73E-04	Lab	Tmm	Stone, 1975
C-1	1.70E-08	4.82E-05	Lab	Tmm	Stone, 1975
C-1	2.23E-07	6.32E-04	Remold	Tmm	Stone, 1975
C-3	4.53E-08	1.29E-04	Remold	Tmm	Stone, 1975
C-3	1.10E-08	3.12E-05	Lab	Tmm	Stone, 1975
C-3	2.50E-08	7.09E-05	Lab	Tmm	Stone, 1975
M62B (RFBL3)	4.57E-07	1.30E-03	Packer	Tmm	Herzog, 1991a
M62B (RFBL3)	6.16E-08	1.75E-04	Packer	Tmm	Herzog, 1991a
M62B (RFBL3)	6.47E-08	1.83E-04	Packer	Tmm	Herzog, 1991a
M32B	4.12E-03	1.17E+01	Slug	Tmm	Kleinfelder, 1988
M34B	2.79E-03	7.91E+00	Slug	Tmm	Kleinfelder, 1988
M39A	4.50E-03	1.28E+01	Slug	Tmm	Kleinfelder, 1988
M40A	1.03E-03	2.92E+00	Slug	Tmm	Kleinfelder, 1988
RFB10	1.10E-06	3.12E-03	Lab	Tmm	Herzog, 1991a
RFB12	6.30E-07	1.79E-03	Packer	Tmm	Herzog, 1991a
RFB12	1.54E-06	4.37E-03	Packer	Tmm	Herzog, 1991a
RFB12	2.91E-07	8.25E-04	Packer	Tmm	Herzog, 1991a
RFB12	7.23E-08	2.05E-04	Lab	Tmm	Herzog, 1991a
RFB15	4.40E-08	1.25E-04	Lab	Tmm	Herzog, 1991a
RFB32	8.65E-07	2.45E-03	Packer	Tmm	Herzog, 1991a
RFB6	1.61E-07	4.56E-04	Packer	Tmm	Herzog, 1991a
RFB6	1.05E-07	2.98E-04	Packer	Tmm	Herzog, 1991a
RFB7	2.63E-07	7.46E-04	Packer	Tmm	Herzog, 1991a
RFB7	9.77E-06	2.77E-02	Lab	Tmm	Herzog, 1991a
RFB7	1.21E-07	3.43E-04	Packer	Tmm	Herzog, 1991a
M59B(RFB27)	2.30E-07	6.52E-04	Lab	Tmm	Herzog, 1991b
M59B(RFB27)	2.29E-07	6.49E-04	Lab	Tmm	Herzog, 1991b
M59B(RFB27)	3.26E-07	9.24E-04	Lab	Tmm	Herzog, 1991b
M59B(RFB27)	7.33E-08	2.08E-04	Lab	Tmm	Herzog, 1991b
RFB32	4.90E-05	1.39E-01	Lab	Tmm	Herzog, 1991b
RFB32	7.76E-06	2.20E-02	Lab	Tmm	Herzog, 1991b
RFB32	6.01E-06	1.70E-02	Lab	Tmm	Herzog, 1991b
RFB32	2.25E-06	6.38E-03	Lab	Tmm	Herzog, 1991b
M62B(RFBL3)	9.83E-07	2.79E-03	Lab	Tmm	Herzog, 1991b
M62B(RFBL3)	3.54E-07	1.00E-03	Lab	Tmm	Herzog, 1991b
M62B(RFBL3)	1.59E-07	4.51E-04	Lab	Tmm	Herzog, 1991b
M62B(RFBL3)	4.84E-07	1.37E-03	Lab	Tmm	Herzog, 1991b
M65B (AB3)	1.02E-04	2.90E-01	Slug	Tmm	Dames & Moore, 1994
M66B (AB4)	4.59E-05	1.30E-01	Slug	Tmm	Dames & Moore, 1994

TABLE 5.11-2 (CONTINUED - Page 3 of 4)

HYDRAULIC CONDUCTIVITY DATA FOR THE PVLF AREA

PALOS VERDES LANDFILL - REMEDIAL INVESTIGATION REPORT

BORING OR WELL	K (cm/sec)	K (ft/day)	TEST TYPE	GEOLOGIC UNIT	DATA SOURCE
M33B	1.59E-03	4.51E+00	Slug	Tmm/Tmv	Kleinfelder, 1988
M24A	1.30E-04	3.69E-01	Slug	Tmv	Kleinfelder, 1988
M42A	4.38E-04	1.24E+00	Slug	Tmv	Kleinfelder, 1988
M43A	2.28E-03	6.46E+00	Slug	Tmv	Kleinfelder, 1988
RFB11	1.06E-07	3.00E-04	Packer	Tmv	Herzog, 1991a
RFB11	1.65E-07	4.68E-04	Packer	Tmv	Herzog, 1991a
RFB11	9.14E-07	2.59E-03	Packer	Tmv	Herzog, 1991a
M53B (RFB16)	1.49E-07	4.22E-04	Packer	Tmv	Herzog, 1991a
RFB19	1.79E-04	5.07E-01	Packer	Tmv	Herzog, 1991a
RFB19	1.10E-07	3.12E-04	Packer	Tmv	Herzog, 1991a
M56B (RFB24)	1.52E-06	4.31E-03	Packer	Tmv	Herzog, 1991a
RFB30	4.47E-06	1.27E-02	Packer	Tmv	Herzog, 1991a
RFB30	6.55E-06	1.86E-02	Packer	Tmv	Herzog, 1991a
RFB32	2.33E-07	6.60E-04	Packer	Tmv	Herzog, 1991a
RFB32	5.07E-07	1.44E-03	Packer	Tmv	Herzog, 1991a
RFB7	6.97E-08	1.98E-04	Packer	Tmv	Herzog, 1991a
RFB7	1.97E-07	5.58E-04	Packer	Tmv	Herzog, 1991a
RFB19	3.32E-05	9.41E-02	Lab	Tmv	Herzog, 1991b
RFB19	1.88E-05	5.33E-02	Lab	Tmv	Herzog, 1991b
RFB19	4.90E-06	1.39E-02	Lab	Tmv	Herzog, 1991b
RFB19	1.32E-06	3.74E-03	Lab	Tmv	Herzog, 1991b
RFB19	5.77E-07	1.64E-03	Lab	Tmv	Herzog, 1991b
M56B(RFB24)	1.13E-06	3.20E-03	Lab	Tmv	Herzog, 1991b
M56B(RFB24)	1.06E-05	3.01E-02	Lab	Tmv	Herzog, 1991b
RFB30	4.91E-07	1.39E-03	Lab	Tmv	Herzog, 1991b
RFB30	4.56E-07	1.29E-03	Lab	Tmv	Herzog, 1991b
RFB32	2.06E-07	5.84E-04	Lab	Tmv	Herzog, 1991b
RFB32	3.42E-06	9.70E-03	Lab	Tmv	Herzog, 1991b
RFB32	3.90E-07	1.11E-03	Lab	Tmv	Herzog, 1991b
RFBL1	1.33E-05	3.77E-02	Lab	Tmv	Herzog, 1991b

TABLE 5.11-2 (CONTINUED - Page 4 of 4)

HYDRAULIC CONDUCTIVITY DATA FOR THE PVLF AREA

PALOS VERDES LANDFILL - REMEDIAL INVESTIGATION REPORT

BORING OR WELL	K (cm/sec)	K (ft/day)	TEST TYPE	GEOLOGIC UNIT	DATA SOURCE
M45A	1.30E-03	3.69E+00	Slug	Tma	Kleinfelder, 1988
M47B	3.70E-04	1.05E+00	Slug	Tma	Kleinfelder, 1988
RFB1	2.00E-05	5.67E-02	Packer	Tma	Herzog, 1991a
RFB1	9.53E-05	2.70E-01	Packer	Tma	Herzog, 1991a
RFB1	1.24E-04	3.51E-01	Packer	Tma	Herzog, 1991a
RFB22	2.09E-07	5.92E-04	Packer	Tma	Herzog 1991a
RFB22	3.64E-07	1.03E-03	Packer	Tma	Herzog, 1991a
RFB22	1.08E-06	3.06E-03	Packer	Tma	Herzog, 1991a
M56B (RFB24)	1.67E-06	4.73E-03	Packer	Tma	Herzog, 1991a
M57B (RFB25)	1.10E-06	3.12E-03	Packer	Tma	Herzog, 1991a
M57B (RFB25)	1.45E-05	4.11E-02	Packer	Tma	Herzog, 1991a
M57B (RFB25)	4.22E-07	1.20E-03	Packer	Tma	Herzog, 1991a
M60B (RFB29)	7.18E-07	2.04E-03	Packer	Tma	Herzog, 1991a
M60B (RFB29)	1.43E-04	4.05E-01	Packer	Tma	Herzog, 1991a
M60B (RFB29)	2.36E-04	6.69E-01	Packer	Tma	Herzog, 1991a
M54B(RBF20)	8.30E-08	2.35E-04	Lab	Tma	Herzog, 1991b
M55B(RBF21)	1.19E-06	3.37E-03	Lab	Tma	Herzog, 1991b
RFB22	8.23E-09	2.33E-05	Lab	Tma	Herzog, 1991b
RFB23	3.12E-08	8.85E-05	Lab	Tma	Herzog, 1991b
RFB23	1.20E-06	3.40E-03	Lab	Tma	Herzog, 1991b
M56B(RBF24)	1.72E-06	4.88E-03	Lab	Tma	Herzog, 1991b
M57B(RBF25)	2.51E-07	7.12E-04	Lab	Tma	Herzog, 1991b
RFB28	2.87E-06	8.14E-03	Lab	Tma	Herzog, 1991b
RFB28	8.52E-06	2.42E-02	Lab	Tma	Herzog, 1991b
RFB28	7.56E-06	2.14E-02	Lab	Tma	Herzog, 1991b
M60B(RFB29)	3.25E-06	9.21E-03	Lab	Tma	Herzog, 1991b
M60B(RFB29)	4.67E-08	1.32E-04	Lab	Tma	Herzog, 1991b
M60B(RFB29)	3.13E-05	8.87E-02	Lab	Tma	Herzog, 1991b
M60B(RFB29)	4.40E-07	1.25E-03	Lab	Tma	Herzog, 1991b
M60B(RFB29)	1.34E-08	3.80E-05	Lab	Tma	Herzog, 1991b
RFB30	7.64E-04	2.17E+00	Lab	Tma	Herzog, 1991b
RFB30	3.44E-06	9.75E-03	Lab	Tma	Herzog, 1991b
RFB30	5.78E-04	1.64E+00	Lab	Tma	Herzog, 1991b
RFBL2	7.84E-08	2.22E-04	Lab	Tma	Herzog, 1991b

Notes:

Qo = Quaternary overburden deposits and landfill refuse
Qus = Quaternary undifferentiated sand deposits
Tmm = Malaga Mudstone member of the Monterey Formation
Tmv = Valmonte Diatomite member of the Monterey Formation
Tma = Altamira Shale member of the Monterey Formation
Bedrock = Monterey, Undifferentiated
cm/sec = centimeters per second
ft/day = feet per day. To convert cm/sec to ft/day, multiply cm/sec by 2,835.
2.36E-04 is scientific notation for 0.000236
Remold K Values are an Average of 85%, 90% and 95% Compactions.

TBL511_2.WP

TABLE 5.11-3

COMPARISON OF HYDRAULIC CONDUCTIVITY VALUES FROM NEW WELLS TO PREVIOUS VALUES

PALOS VERDES LANDFILL - REMEDIAL INVESTIGATION REPORT

FORMATION TESTED	PREVIOUS LOW VALUE	PREVIOUS HIGH VALUE	NEW WELLS' LOW VALUE*	NEW WELLS' HIGH VALUE*
Qus	1.02×10^{-2}	5.95	18.98	19.78
Tmm	3.12×10^{-5}	7.91	0.13	0.29
Qo/Tmm	3.97×10^{-2}	3.40	0.62	9.91
Qus/Tmm	none tested	none tested	0.37	5.14

TBL511_3.WP

* The Cooper, Bredehoeft, Papadopulous derived value was used, which gives transmissivity. To determine the hydraulic conductivity, the transmissivity was divided by the aquifer thickness which was assumed to be the distance from the water table in the well to the bottom of the filter pack. If the true aquifer thickness is greater, as is expected at M67B (formerly AB8) and M68B (formerly AB9), then the hydraulic conductivity value would be smaller.

The previous aquifer tests included both laboratory and field tests.

Qo = Quaternary overburden deposits (colluvium, alluvium, fill, etc.).

Qus = Quaternary undifferentiated sand deposits (Lomita Marl, San Pedro Sand, etc.).

Tmm = Malaga Mudstone member of the Tertiary-aged Monterey formation.

Hydraulic conductivity (K) units are in feet per day.

instead of 19 ft/day. The values of hydraulic conductivity derived from the Hvorslev and Bouwer and Rice methods, which are about one-third of the values derived from the CBP method, support this argument for lower values. The range of hydraulic conductivity values from these methods for M67B (AB8) and M68B (AB9) is 5.08 to 6.13. These values are very close to the high value from previous studies.

The hydraulic conductivity values for the Tmm flow zone from the additional wells (M65B (AB3) and M66B (AB4)) fall within the range of previously tested data. The range of hydraulic conductivity values for the Tmm flow unit is quite large, which is not unexpected because of the fractured nature of the Malaga Mudstone. The number of water bearing fractures which are intersected by the screened portions of the wells in this formation will greatly influence the local hydraulic conductivity values.

The hydraulic conductivity results for the two new wells screened across both the Qo and the Tmm flow zones (M69B (AB6) and M70B (AB7)) were on the high end of the previous results. M69B (AB6) is slightly higher than the previous high result, and the value may be larger than the calibrated model value by two orders of magnitude in some areas. However, this was addressed by the uncertainty analysis in the flow models (described previously). The percentage of screen and water adjacent to the Qo versus the Tmm and the number of water bearing fractures in the Tmm flow zone would influence the test results, and accounts for the wide variations in test results.

Previous hydraulic conductivity test results were unavailable for wells screened across both the Qus and Tmm flow zones to compare against the new wells M63B (AB2) and M64B (AB1a). The values found during the additional downgradient investigations for these flow zone wells, however, do fall within the previous ranges for both Qus and Tmm separately. Based on this comparison, it appears that the results from the additional downgradient hydrogeologic field program are reasonably consistent with values found during previous investigations for the flow zones tested.

Effective Porosity Values

The new data set includes four porosity values determined in the laboratory; two for the San Pedro Sand and two for the Malaga Mudstone. These values are listed below.

San Pedro Sand:	0.399, 0.516
Malaga Mudstone:	0.606, 0.614

The above values can be only compared with the chemical transport model's effective porosity values in a qualitative sense. In reality, owing to the presence of dead-end pores, effective porosity tends to be somewhat smaller than total porosity. The values of effective porosity used in the transport model range between 0.005 to 0.3 in the vicinity of the PVLF, and 0.3 to 0.4 in the West Coast Basin. For the same flux (flow) and hydraulic conductivity, the smaller the effective porosity, the faster the ground water flow and chemical transport. The model may therefore be considered conservative, because the higher values found during the additional downgradient hydrogeologic field program, if used, would slow down chemical transport from the PVLF.

Water Quality Data

The new water quality data (the ensuing discussion is limited to some organic chemicals which are likely to be anthropogenic, e.g., tetrachloroethylene, trichloroethylene, benzene, vinyl chloride, etc.) indicate that, in general, the chemical concentrations in the new monitoring wells are consistent with the transport model, except for the abnormally high concentrations of benzene in M67B (AB8) and M68B (AB9) on the West Coast Basin side of the fault zone. Because these concentrations are much greater than benzene concentrations observed in wells on the PVLF side of the fault zone, the benzene found in M67B (AB8) and M68B (AB9) may have originated from a source in the West Coast Basin. In addition, the benzene was only detected on the initial sampling episode following well installation, but not since that time. The water quality data from the additional remedial investigations are discussed in more detail in Section 3.6.

5.11.4 Uncertainty Scenarios

The new data set highlights uncertainties relating to the following:

- Hydrogeologic functions of the fault;
- Hydrogeologic parameters between the PVLF and the fault;

- Local ground water flow regime between the PVLF and the fault; and,

- Vertical hydraulic gradient.

It is necessary to keep in mind that the flow and transport models were developed for the purposes of risk assessment. Many of the uncertainties were overcome through the incorporation of conservative assumptions in order to over predict contaminant concentrations at potential receptors. The uncertainties listed above are discussed in terms of their impact on chemical transport and prediction of contaminant concentrations at potential receptors.

Hydrogeologic Functions of the Fault

The effects due to the hydrogeologic functions of the fault have been evaluated through sensitivity analyses. The following analyses were conducted.

In the first ground water flow model (prior to developing the chemical transport model) the following sensitivity analyses were included:

a. Creation of "holes" in the fault zone by increasing the hydraulic conductivity value by 1,000 fold in selected locations along the fault axis (Sensitivity Analysis Case 8).

b. Increasing the hydraulic conductivity values in the zone immediately upgradient of the fault zone to study potential flow deflection by the fault (Sensitivity Analysis Case 13).

c. Hydraulic conductivity values in the fault zone were assigned a minimum value of 1.0×10^{-5} cm/sec (0.03 ft/day), thereby eliminating the effect of the fault as a flow barrier (Sensitivity Analysis Case 16).

Results from the above sensitivity analysis cases indicated that the bulk of the ground water (represented by fluid particles) may discharge into the West Coast Basin more easily. The variation of the fault functions could also affect the flow upgradient from the fault zone. However, it was noted that the hydraulic gradient across the PVLF area was not significantly affected by the fault function variability. This observation suggested that the accuracy in determining the chemical flux leaving the PVLF would not be adversely impacted by uncertainty relating to the fault function, provided that the hydraulic conductivity values are approximately correct.

In the second ground water flow model (after re-calibration for matching heads and chemical concentrations) the following sensitivity analysis was added:

a. The fault in Layers 1 and 2 was partially removed (Sensitivity Analysis Case 9).

Results from the above sensitivity analysis case indicated that the partial removal of the fault zone caused the maximum chemical concentration (vinyl chloride) to be elevated at all potential receptors by less than an order of magnitude. Dames & Moore's reports on both the ground water and contaminant flow modeling are attached as Appendices E.2 and E.3.

Hydrogeologic Parameters Between the PVLF and the Fault Zone

The effects due to uncertainty relating to hydrogeologic parameters between the PVLF and the fault zone has been indirectly evaluated through sensitivity analyses of basin-wide variation of hydrogeologic parameters. The following analyses were conducted.

In the first ground water flow model the following conditions were investigated:

a. Increasing basin-wide hydraulic conductivity sensitivity analysis cases indicated relatively insignificant changes in the general flow patterns. However, the velocity of ground water could increase by one order of magnitude.

In the second ground water flow model the specific sensitivity cases included:

a. Effective porosities of all formations were increased by a factor of 2 (Sensitivity Analysis Case 2).

b. Hydraulic conductivities of the Overburden and San Pedro Sand were increased by a factor of 2 (Sensitivity Analysis Case 6).

c. Hydraulic conductivities of the Malaga Mudstone and the Valmonte Diatomite were increased by a factor of 2 (Sensitivity Analysis Case 7).

Results from the above sensitivity analysis cases indicated that the change in the maximum chemical concentration (vinyl chloride) at all potential receptors was less than an order of magnitude.

Local Ground Water Flow Regimes Between the PVLF and the Fault

The effects due to uncertainties associated with local ground water flow regimes were indirectly evaluated through the variation of hydrogeologic parameters and fault hydrogeologic functions, as just described under the two previous headings.

Presence of Vertical Hydraulic Gradient

The effects due to uncertainty relating to the presence of vertical hydraulic gradient have been evaluated through a sensitivity analysis case. The following analysis was conducted with the second ground water flow model.

a. Downward vertical hydraulic gradient was imposed between model Layers 1, 2, and 3 (Sensitivity Analysis Case 5).

Results from the above sensitivity analysis case indicated that the change in the maximum chemical concentration (vinyl chloride) at all potential receptors was less than an order of magnitude.

It should be pointed out here that vertical gradient causes the contaminants to migrate downward into organic carbon-rich Malaga Mudstone. Most of the chemicals of concern are organic chemicals which would be vigorously adsorbed, and therefore severely retarded, by organic carbon in the Malaga Mudstone. Because of the lack of information relating to vertical gradient, the model was designed to be conservative by forcing the flow in the vicinity of the PVLF to occur mainly in the horizontal direction. Thus, significant loss of contaminants to the Malaga Mudstone was avoided, causing the contaminant concentrations to be overestimated downgradient of the PVLF. Additionally, predominant horizontal flow would tend to cause the chemicals of concern to auras at the potential receptors in a more direct fashion, thus quicker, than if strong vertical gradients caused more vertical flow. Therefore, the model is again shown to be conservative with respect to predicting arrival times and concentrations of chemicals of concern.

5.11.5 Summary

The hydrogeologic data obtained during the additional remedial investigations from northeast of the PVLF were evaluated in the context of the adequacy of the existing modeling results in addressing new technical issues raised by the new data. The following technical issues were discussed: adequacy of the existing conceptual model; agreement between the new data and the developed models; and potential uncertainty scenarios.

The new data raised a number uncertainty issues relating to the conceptual model. These issues were addressed by uncertainty analyses performed during construction of both flow models, and in the conservative design of the models. The models have been developed for the ultimate purposes of risk assessment. A number of conservative assumptions were incorporated into the model to compensate for uncertainties in the hydrogeologic information. Because of the conservativeness of the existing models and the fact that the new data set do not alter the conceptual model significantly, the regeneration of the models for risk assessment purposes appears unnecessary at this time. However, comparisons between the modeling results and the future water quality data should be carried out on a regular and consistent basis. Deviations from the modeled data, in particular contaminant levels from the site monitoring program statistically greater than modeled levels, are indicative that the ground water and chemical transport models should be regenerated with the new monitoring data.