

6.0 CONTAMINANT FATE AND TRANSPORT

This section provides an analysis of the fate and transport of PCE and TCE, the primary contaminants of concern for the Muscoy Plume OU. The analysis identifies the environmental media (i.e., water, air, soil, biota) potentially affected by contaminant migration and the possible extent of the contamination within each media. The results of this analysis will provide a basis for determining potentially exposed receptors and an estimation of the levels of potential exposure.

The fate and transport of chemical compounds released into the environment are influenced by the chemical and physical characteristics of the contaminants, their persistence in the environment, source characteristics, release mechanisms, and the transport mechanisms and contaminant migration pathways.

Although a screening-level evaluation of the fate and potential transport of contaminants within and between other media is presented in this section, the primary focus will be on the fate and transport of PCE and TCE in groundwater. The following subsections discuss the physical and chemical characteristics and environmental persistence of PCE and TCE; source characteristics and the affected environmental media; the transport mechanisms and pathways; an analysis of the nature and extent of groundwater contamination based on the results of the Interim Sampling Event and groundwater modeling; and a screening risk assessment.

6.1 CONTAMINANT CHARACTERISTICS AND ENVIRONMENTAL PERSISTENCE

6.1.1 Chemical and Physical Characteristics

PCE and TCE are chlorinated aliphatic hydrocarbons belonging to a family of unsaturated hydrocarbons known as alkenes. Alkenes are distinguished by the presence of a carbon-carbon double bond, indicated as C=C. At room temperature both PCE and TCE are colorless, volatile, nonflammable, dense liquids with characteristic odors that are mildly sweet and resemble chloroform. Although relatively insoluble in water, PCE and TCE are miscible with alcohol, ether, chloroform, and benzene (Merck 1989). The physical properties of PCE and TCE are provided in Table 6-1.

PCE is used as a solvent in a wide variety of industrial and commercial applications and products including dry cleaning, degreasing, paints and coatings, adhesives, and as a registered pesticide for controlling wasps and hornets (CARB 1991). TCE has been used in degreasing operations, polyvinyl chloride (PVC) production, adhesive formulations, painting and coating operations, as a refrigerant and heat exchange liquid (CARB 1990), and was also used in the past as an anesthetic.

To describe the persistence of common groundwater contaminants, an EPA groundwater supply survey of 466 randomly selected public groundwater supply systems detected one or more VOCs in groundwater samples collected from 16.8% of small water systems and 28% of large supply systems. The two VOCs most commonly present were PCE and TCE (EPA 1987).

Table 6-1

PHYSICAL PROPERTIES OF PCE AND TCE

Property	PCE	TCE	Reference
Synonyms	Tetrachloroethene (IUPAC), Perchloroethylene, Ethene, Tetrachloroethylene, Perk Ethylene Tetrachloride, Tetra Cap, Perclene	Trichloroethene (IUPAC) Ethylene Trichloride, Triclene, Trilene, Algylen, Chlorylen, Gemalgene, Trethylene, Trichloran	Weiss 1986, Keller 1992
Chemical Formula	Cl ₂ C = CCl ₂	Cl ₂ C = CHCl	Merck 1989
CAS Registry Number*	127-18-14	79-01-6	Keller 1992
NIOSH RTECS Number	KX3850000	KX4550000	NIOSH 1990
Molecular Weight	165.85	131.40	Merck 1989
Boiling Point (760 mm Hg)	121°C	86.7°C	Merck 1989
Melting Point	-22°C	-84.8°C	Merck 1989
Water Solubility (20°C)	400 mg/l 150 mg/l	1000 mg/l 1100 mg/l	Lyman 1990 EPA 1986b
Specific Gravity (20°C)	1.62	1.46	Merck 1989
Vapor Pressure (20 to 30°C)	17.8 mm Hg	57.9 mm Hg	EPA 1986b
Henry's Law Constant (atm-m ³ /mol)	0.0259 0.0149 0.0083	0.00910 0.0103 0.01	EPA 1986b Howard 1990 Lyman 1990
Organic Carbon Partition Coefficient (K _{oc})	282.75 ml/g	147.48 ml/g	Hassett et al 1983*
Log Octanol/Water Partition Coefficient (Log K _{ow})	2.60	2.29	Vershueren 1983
Conversion Factor: Gas Phase (25°C, 760 mm Hg)	1 ppbv = 6.78 ug/m ³ 1 ug/m ³ = 0.15 ppbv	1 ppbv = 5.37 ug/m ³ 1 ug/m ³ = 0.19 ppbv	
Diffusion Coefficient (30°C)	.07852 cm ² /sec	.08606 cm ² /sec	EPA 1988c

IUPAC = International Union of Pure and Applied Chemistry (nomenclature system)

CAS = Chemical Abstracts Service Registry Number*

NIOSH RTECS = National Institute for Occupational Safety and Health Registry of Toxic Effects of Chemical Substances

* Calculated from the relationship: Log K_{oc} = 0.909Log K_{ow} + 0.088

1 **6.1.2 Contaminant Fate and Persistence**

2 As discussed in Subsection 5.4, aerobic conditions are expected to predominate in this aquifer. Only
3 localized areas of anaerobic conditions might occur. There has been no observed indication of a
4 biological oxygen demand source which would affect the general redox conditions of the aquifer and no
5 primary substrate for biological activity. Natural sources of petroleum hydrocarbons may be present,
6 however, in the groundwater basin.

7 **Tetrachloroethene (PCE)**

8 When released to surface soils, particularly dry soil, PCE is subject to fairly rapid evaporation due to
9 various chemical parameters, such as its high vapor pressure and Henry's Law constant, and low soil
10 adsorption (K_{oc}). The importance of these parameters is discussed in Subsection 6.3.1. PCE exhibits low
11 to medium mobility in soil but can percolate fairly rapidly through dry, sandy soil to reach underlying
12 groundwater.

13 Although stable under aerobic conditions, PCE can undergo slow biotransformation by reductive
14 dechlorination under anaerobic methanogenic conditions through a process of transformation known as
15 co-metabolism. The process is mediated by the presence of a primary substrate (e.g., petroleum
16 hydrocarbons) for energy and growth, and the conditions necessary for its production by acclimated
17 microorganisms such as methane-producing anaerobes (methanogens) or sulfate-producing bacteria. The
18 reaction involves a sequential reductive dehalogenation (e.g., PCE to TCE, TCE to the dichloroethylene
19 isomers cis- and trans- 1,2 dichloroethylene, vinyl chloride, and finally ethylene and ethane [Freedman
20 and Gosset 1989]). The mechanism consists of the chlorinated compounds accepting electrons from other
21 organic compounds, or geologic material in the aquifer, and the subsequent release of chlorine ion (Cl⁻).
22 The process is slow and normally requires long lag periods before active transformation begins (EPA
23 1987; Howard 1990). A more thorough discussion of the process is presented in Section 6.3.1. In
24 general, the rates of the reaction tend to be higher for the more chlorinated compounds such as PCE and
25 TCE.

26 The persistence and fate of PCE vapor in the atmosphere is predominantly affected by chemical removal
27 mechanisms, consisting of ultraviolet catalysis, or photo-chemical reactions with atmospheric oxidants
28 (hydroxyl [OH] radical, ozone [O₃], or nitrate [NO₃] radical), principally the OH radical.

29 At average atmospheric (i.e., tropospheric) temperature and OH concentration, the atmospheric lifetime
30 of PCE is expected to be approximately 150 days (CARB 1991). Other estimates of PCE's persistence
31 in the atmosphere range from a half-life of approximately 60 days to complete degradation in an hour
32 (Howard 1990). Physical removal mechanisms (rain washout, dry deposition, adsorption on aerosols)
33 are of negligible importance because of PCE's physical properties (e.g., polarity, solubility, adsorptivity,
34 vapor pressure) and the long removal times associated with these mechanisms (CARB 1991).

35 When released to surface water, PCE is subject to rapid volatilization with an estimated half-life ranging
36 from less than one day to several weeks. Biodegradation, bioconcentration in aquatic organisms or
37 adsorption to sediments are not considered significant (Howard 1990).

1 **Trichloroethene (TCE)**

2 TCE is highly volatile. When released to soil, some TCE will volatilize and transfer to the atmosphere.
3 The liquid phase, being dense, with a low tendency to adsorb onto soil (i.e., K_{oc}), will percolate
4 downward to underlying groundwater.

5 Biodegradation in surface water is extremely slow under most conditions. Some studies have noted
6 significant aerobic biodegradation, while others found no biodegradation in screening or seawater studies
7 (Howard 1990).

8 Anaerobic transformation, or biodegradation, studies using aquifer material known to support
9 methanogenesis, resulted in the removal of 70-99% of the TCE after 40 weeks (Howard 1990). The
10 reductive co-metabolic process, discussed previously for PCE, involves a slow sequential reductive
11 dehalogenation to the dichloroethylene isomers, vinyl chloride, and then to ethylene and ethane. TCE,
12 unlike PCE, may also undergo co-metabolic oxidative dehalogenation by methanotrophic cultures with
13 the possible production of glyoxylic acid and dichloroacetic acid intermediates (McCarty 1988).

14 The persistence of TCE in the atmosphere is predominantly affected by removal through photo-chemical
15 reactions with highly reactive radicals (e.g., hydroxyl [OH] radical, ozone [O₃], or nitrate [NO₃] radical).
16 The principal mechanism involves reactions with OH radicals during daylight hours with the formation
17 of formyl chloride, phosgene, and, possibly, dichloroacetyl chloride (CARB 1990). Ozone and NO₃
18 radical reactions take too long to compete with the OH radical reaction and, therefore, are of minor
19 importance. TCE is moderately persistent, with an atmospheric lifetime of 4 to 15 days (CARB 1990;
20 Howard 1990). Physical removal mechanisms (rain washout, dry deposition, adsorption on aerosols) are
21 considered negligible factors influencing its atmospheric persistence.

22 When released to surface water, TCE is subject to rapid volatilization with an estimated half-life ranging
23 from less than one day to several weeks. Biodegradation, bioconcentration in aquatic organisms or
24 adsorption to sediments are not considered significant (Howard 1990).

25 **6.2 AFFECTED ENVIRONMENTAL MEDIA**

26 This subsection describes the distribution of the contaminants in the affected environmental media within
27 the Muscoy Plume OU. The Newmark OU RI/FS Report (Section 6.2) discussed characteristics of the
28 suspected source area at the time of that RI/FS. As discussed in Section 1.0 of this Muscoy Plume RI/FS
29 Report, the focus of the Muscoy Plume OU RI/FS is on groundwater contamination in the leading portion
30 of the contaminant plume in the Muscoy area. Discussion of potential source areas are therefore not
31 contained within the current project but will be addressed during the Source OU RI/FS.

32 **6.2.1 Affected Environmental Media**

33 Substances released at or slightly below land surface enter groundwater through percolation or direct
34 migration. Direct migration can occur through leakage from sources lying within the saturated zone
35 (e.g., storage tanks, pipelines, wells). Solid, soluble contaminants dissolved in precipitation, runoff, or
36 applied water can migrate through percolation into the soil. The rate of movement depends on the water
37 recharge rates (e.g., infiltration of rainwater through a contaminated soil zone) and contaminant solubility.

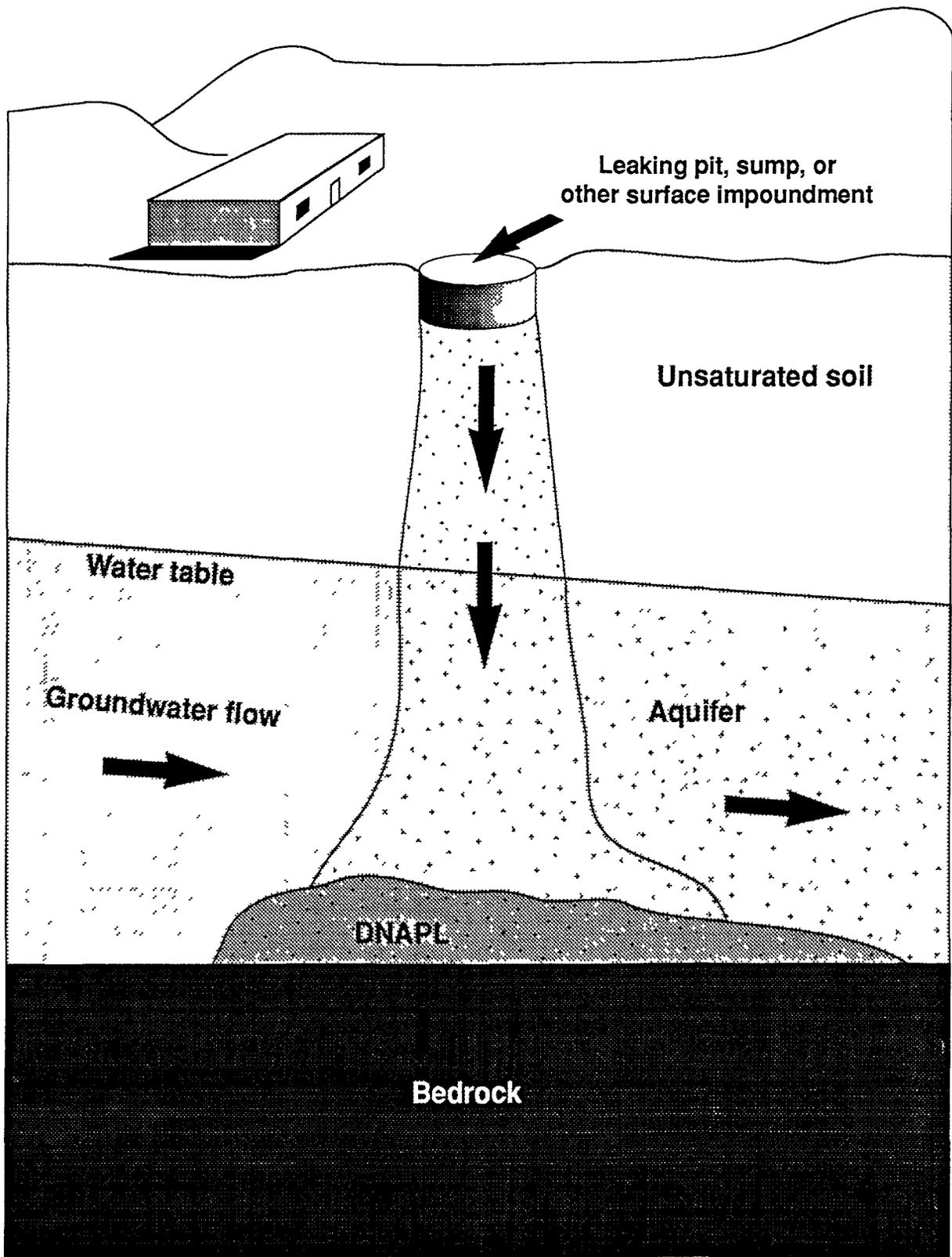
1 Liquid organic contaminants such as PCE and TCE can percolate directly into soils. Figure 6-1 provides
2 a simplified schematic of a TCE or PCE release. The percolating liquid or leachate continues to migrate
3 downward into the saturated zone then spreads vertically and horizontally following the pattern of
4 groundwater flow. Additionally, groundwater contamination can result from hydraulic mixing, or
5 interaquifer exchange, with a contaminated aquifer or through groundwater recharge from a contaminated
6 surface water body (EPA 1987).

7 Organic liquids of moderate to low solubilities such as PCE (150-400 mg/l) and TCE (1000-1100 mg/l)
8 can contaminate as much as 10,000 times their own volume, up to 100% of their solubilities. The
9 concentrations of these contaminants in groundwater, however, are only rarely close to their solubility
10 limits, even when their solubilities are increased by the presence of other solvents or when they are
11 present in the aquifer as a nonaqueous phase liquid (NAPL). The concentrations observed in the
12 environment are generally an order of magnitude lower than the contaminant's solubility in water. This
13 is due to dilution of the contaminant through dispersion and spreading in the groundwater and the
14 relatively slow kinetics of chemical desorption from soil. Consequently, numerous or continuous small
15 spills or leaks of tens of gallons of solvent that may have been considered insignificant at the time could
16 constitute a significant contaminant source if the organic liquid reached the groundwater (Mackay et al.
17 1985). A more detailed discussion of the factors influencing contaminant movement is provided in
18 Section 6.3.

19 The volume of groundwater that could be contaminated by an organic NAPL layer is substantial. In
20 addition, the temporal extent of the contamination would be quite large since substantial time would have
21 to pass before the flowing groundwater could exhaust the supply of the contaminant(s) contained in the
22 NAPL layer.

23 As depicted in Figure 6-1, NAPLs with densities greater than water, such as PCE and TCE, can sink
24 through the saturated zone until an impermeable layer, or confining bed, is encountered. The movement
25 of dense NAPLs (DNAPLs) will follow the slope of the impermeable boundary while a fraction of the
26 DNAPL will dissolve and move with groundwater flow.

27 The presence of a DNAPL is generally indicated when the dense organic liquid contaminant
28 concentrations approach 1% of the contaminant's solubility limit. However, given the solubilities of
29 PCE (150 to 400 mg/l) and TCE (1,000 to 1,100 mg/l), groundwater concentrations in samples taken
30 from monitoring wells or municipal supply wells would have to approach 1,500 $\mu\text{g}/\ell$ for PCE or 10,000
31 $\mu\text{g}/\ell$ for TCE to suggest the presence of a nearby DNAPL layer. The highest concentrations of
32 contaminants were 27 $\mu\text{g}/\ell$ for PCE and 6 $\mu\text{g}/\ell$ for TCE (see Table 5-1). In addition, considerable
33 variability in contaminant concentrations would be expected if a DNAPL were present. However, there
34 was little variability in the PCE and TCE concentrations observed in groundwater samples collected from
35 municipal supply wells in the Muscoy Plume OU. Consequently, the presence of DNAPLs, at least in
36 the area of the plume downgradient of the suspected source area monitoring wells, is not considered
37 likely.



Adapted from Fetter 1988 (Figure 10.20)

Figure 6-1
 TCE/PCE Release
 Muscoy Plume OU RI/FS Report
 Newmark Groundwater Contamination Superfund Site

LEGEND	
DNAPL	= Dense non-aqueous phase liquid

6.3 TRANSPORT MECHANISMS AND PATHWAYS

6.3.1 Transport Mechanisms

The transport or migration of contaminants within the groundwater and into other environmental media (surface water, air, soil, biota), is affected by a number of transport mechanisms and environmental factors. These mechanisms are related to the contaminants' physicochemical properties, including:

- Solubility in water;
- Specific gravity;
- Volatility (e.g., vapor pressure);
- Henry's Law constant and other partition coefficients that define the distribution of chemicals between two phases; and
- The processes that govern the movement of contaminants in groundwater (advection, dispersion, retardation, transformation).

These properties are briefly described below.

Solubility

The solubility of a chemical is defined as the maximum amount of a chemical that will dissolve in pure water at a specified temperature, pressure, and pH. Solubility is a controlling factor affecting how a contaminant is distributed in surface and groundwater. Highly soluble chemicals tend to remain dissolved in the water column, can be rapidly leached from contaminated soil, and are generally quite mobile in groundwater. Soluble chemicals also tend to be more readily biodegradable than those with low solubility. As shown in Table 6-1, PCE and TCE are slightly to moderately soluble in water and are considered moderately mobile.

It should be noted that the solubilities of hydrophobic organics, such as PCE and TCE, increase in the presence of a polar organic solvent, or cosolvent, in the groundwater. This is known as a cosolvent effect. Furthermore, since sorption and solubility tend to be inversely related, any increase in the solubility of the hydrophobic organic would also result in a proportional decrease in sorption.

Solubility can also affect the rate of dissolution of a NAPL in an aquifer. As groundwater moves past an NAPL (e.g., PCE, TCE), some solvent will dissolve from the surface of the NAPL and diffuse into the groundwater. In time, the groundwater will remove the solvent. However, in relatively stagnant or low-velocity aquifers, an equilibrium concentration approaching the solvent solubility limit may be established in the groundwater, resulting in very little dissolution of the NAPL. The rate of dissolution depends upon a number of factors including:

- Aquifer characteristics (e.g., groundwater flow rate, hydraulic gradient; permeability, organic carbon content, fractures, etc.);
- Properties of the solvent (e.g., solubility, vapor pressure, sorption potential, etc.);
- Thickness of the NAPL;
- Interactions with other contaminants (e.g., cosolvent effects, chemical reactions, etc.); and
- Other processes (e.g., biotransformation, contaminant/aquifer interaction).

1 It should be noted that, based on the project flow model, flow velocities in the Muscoy Plume OU
2 investigation area are high and, except for localized stagnant or low flow areas, contaminant
3 concentrations might be expected to be much less than equilibrium.

4 **Specific Gravity**

5 Specific gravity is a unitless value that defines the ratio of the density (e.g., weight per unit volume
6 expressed in g/cm^3 or g/ml) of a substance (at a given temperature, commonly 20°C) to the density of
7 water at the temperature of its maximum density (1 g/cm^3 at 4°C). Consequently, if the specific gravity
8 of a substance is less than 1, it will float in water, and is termed a LNAPL; substances having specific
9 gravities greater than 1, such as PCE and TCE, will sink in water and are termed DNAPLs.

10 **Volatility**

11 Volatility, the tendency of a substance to change from the liquid phase to the gaseous phase, is an
12 important parameter to consider when evaluating potential air emissions from a site. Volatilization occurs
13 when molecules of a dissolved or pure substance escape to the ambient air or adjacent gas layer. The
14 relative volatility of a substance is expressed in terms of its vapor pressure. Vapor pressure is the
15 pressure (measured in pounds per square inch [psi] or millimeters [mm] of mercury [Hg]) exerted by a
16 chemical's vapor when in equilibrium with its liquid or solid phase. Vapor pressure is a good indicator
17 of the potential of a chemical to volatilize from land. Chemicals with vapor pressures greater than 10
18 mmHg, such as PCE and TCE, are considered highly volatile.

19 The volatilization rate at the soil/air interface is dependent on the chemical's concentration and physical
20 properties (e.g., solubility, specific gravity, vapor pressure), as well as the properties of the soil (e.g.,
21 moisture, temperature, clay and organic content) and adjacent gas layer (e.g., temperature, humidity,
22 wind speed). The rate of volatilization is dependent on the chemical's Henry's Law Constant discussed
23 below, and the chemical concentration and extent of mixing and replenishment in the liquid boundary
24 layer of the liquid/air interface (discussed below).

25 **Henry's Law Constant**

26 Vapor pressure, solubility, and molecular weight are factors that control volatilization of organic
27 chemicals from water to air. All of these factors are reflected in Henry's Law Constant, H. H is actually
28 a partition coefficient that provides an indication of the tendency of a chemical to partition between the
29 soil gas phase and the soil water in the vadose or saturated zones. Values for H can be directly measured
30 or calculated from the chemical's water solubility and vapor pressure using the following relationship:

$$31 \quad H \text{ (atm - m}^3\text{/mole)} = \text{vapor pressure (atm)} \times \text{MW (g/mole)/water solubility (g/m}^3\text{)} \quad (6.1)$$

32 The nondimensional Henry's Law Constant, or H' , is a distribution coefficient that predicts the
33 distribution or ratio at equilibrium of the concentration of a chemical in the gas phase (C_a) to its
34 concentration in the liquid or water phase (C_w) (i.e., $C_a = H' C_w$). Values for H' may be obtained from
35 the literature, experimentally determined, or calculated using the following equation:

$$36 \quad H' = H \text{ (atm - m}^3\text{/mole)} / R \text{ (atm - m}^3\text{/mole} \times \text{ }^\circ\text{K)} \times T \text{ (}^\circ\text{K)} \quad (6.2)$$

1 where: H is the dimensional Henry's Law Constant determined using the relationship described in
2 equation (6.1) above; R is the gas constant (8.2×10^{-5} atm - m³/mole x °K); and T is the ambient
3 temperature (°K). H' at a standard temperature of 20 to 25°C is equal to 41.7 x H.

4 Chemicals having H values greater than 10^{-3} atm - m³/mole and vapor pressures greater than 10 mm Hg,
5 such as PCE and TCE, are considered highly volatile chemicals that tend to volatilize fairly rapidly from
6 water.

7 **Partition Coefficients.** Partition coefficients describe the distribution of a chemical contaminant between
8 two different media or phases. The coefficients that are particularly important parameters for assessing
9 groundwater transport pathways include the following:

10 ▪ Octanol/Water Partition Coefficient (K_{ow}). K_{ow} defines the ratio of a chemical's
11 concentration in octanol to that in water. It is representative of a chemical's tendency
12 to partition itself between an organic non-polar phase and an aqueous phase (i.e., its
13 degree of hydrophobicity). K_{ow} is also related to water solubility, soil/sediment
14 adsorption coefficients (see below), and bioconcentration factors for aquatic life.
15 Chemicals with low K_{ow} values (less than 100) are considered relatively hydrophilic and
16 tend to have high water solubilities, small soil/sediment adsorption coefficients, and small
17 bioconcentration factors for aquatic life. Chemicals with high K_{ow} values (e.g., greater
18 than 10,000) are very hydrophobic and tend to move more readily through biological
19 membranes. PCE ($K_{ow} = 398$) and TCE ($K_{ow} = 195$) fall somewhere in between.

20 ▪ Organic Carbon Partition Coefficient (K_{oc}). K_{oc} , a measure of the relative adsorption
21 potential of organic chemicals, indicates the tendency of an organic compound to adsorb
22 onto the organic fraction of soils or sediments. K_{oc} is defined as the ratio at equilibrium
23 of the amount of chemical adsorbed per unit weight of organic carbon (μg solute
24 adsorbed/g organic carbon) to the chemical concentration in solution (μg solute/ml
25 solution). The K_{oc} for organic contaminants is essentially independent of site-specific soil
26 properties. Low K_{oc} values indicate greater soil mobility or faster leaching through the
27 soil, high K_{oc} values indicate tight bonding to soils and less mobility. K_{oc} can be
28 determined from the following relationship taken from Hassett et al. (1983):

29
$$\log K_{oc} \text{ (ml/g)} = 0.909 \log K_{ow} + 0.088 \quad (6.3)$$

30 Values may range from 1 to 10,000,000 ml/g. Based on relatively low K_{oc} values, PCE
31 (283 ml/g) and TCE (148 ml/g) are expected to exhibit low soil adsorption potential and
32 moderate soil mobility.

33 ▪ Soil/Water Distribution Coefficient (K_d). K_d is a measure of the tendency of a chemical
34 to partition between water and adjoining soil or sediments. The simplest and most
35 common relationship for expressing adsorption potential is as follows:

36
$$K_d = C_s/C_w \quad (6.4)$$

37 where: C_s is the concentration in soil ($\mu\text{g/g}$) and C_w is the concentration in water ($\mu\text{g/ml}$)
38 (Dragun 1988).

1 If equation (6.4) is normalized on the basis of site-specific soil organic carbon content,
2 much of the variation observed among K_d values over different soils can be eliminated.
3 Normalized, or site-specific, K_d values can be calculated using the following relationship:

$$4 \quad K_d \text{ (m\ell/g)} = K_{oc} \times f_{oc} \quad (6.5)$$

5 where: K_{oc} (m\ell/g) is measured, or estimated using equation (6.3) or other appropriate
6 estimation methods described in Lyman (1990); and f_{oc} is the percent soil organic carbon
7 content (mg organic carbon/mg soil). Since K_d is equal to the product of the chemical's
8 K_{oc} and the fraction of organic carbon in the site's soil, the value of K_d , unlike K_{oc} , may
9 vary considerably in different soils. The f_{oc} for saturated zone soils (50 feet or more
10 below ground surface) in the study area is estimated to be less than 0.1% (Best 1992;
11 Mackay and Vogel 1985; Karickhoff 1981). Using the conservative estimate of 0.1%,
12 the K_d values for PCE and TCE for study area soils are calculated to be 0.283 m\ell/g and
13 0.148 m\ell/g, respectively.

14 **Groundwater Contaminant Migration**

15 The transport of organic contaminants dissolved in groundwater is governed by advection-dispersion
16 processes (e.g., fluid movement, fluid mixing, diffusion), and the geochemical processes that either result
17 in a loss of the contaminant from the plume (transformation or degradation) or delay its movement
18 (retardation and sorption). These processes are briefly discussed below.

19 Advection, the process of transport resulting from the gross fluid movement of flowing groundwater, is
20 the dominant factor affecting the movement of dissolved contaminants. Groundwater flows from regions
21 of higher water levels to regions where the water level is low, or in the direction of decreasing hydraulic
22 head pressure. The term used to describe the magnitude of this movement is the hydraulic gradient. The
23 velocity of the flow is equal to the product of the hydraulic gradient and the ability of the aquifer medium
24 to transmit water (e.g., ratio of the medium's hydraulic conductivity and porosity). Based on the results
25 of the project flow model discussed in Subsection 6.4.2, the average groundwater velocity in the Muscoy
26 plume area was calculated to be 500 ft/yr.

27 Dispersion is the processes of molecular diffusion, and mechanical mixing resulting from variations in
28 groundwater velocity (i.e., frictional forces, varying pore sizes, varying path length, velocity gradient
29 variations across the pore space, and the splitting of flow around soil particles), that cause dissolved
30 contaminants to spread as they move with the groundwater. Dispersion and spreading result in
31 contaminant dilution and reduction of contaminant peaks. Consequently, maximum concentrations tend
32 to diminish with increasing distance from the source.

33 Retardation and sorption are the chemical and physical processes that slow or delay the movement of
34 contaminants in groundwater. Some dissolved contaminants may adsorb onto the aquifer material
35 resulting in reduced aqueous phase concentrations, and effectively retarding contaminant movement
36 relative to groundwater flow. For halogenated hydrocarbons such as PCE and TCE, adsorption is
37 affected by the contaminants' hydrophobicity and tendency to adsorb onto organic carbon in the aquifer
38 soils or solids (i.e., soil/water distribution coefficient, or K_d).

1 In a homogeneous aquifer, a linearly retarded contaminant would move at a constant average velocity (V_c)
2 equal to the groundwater's average velocity (V_w) divided by the retardation factor, R_f . R_f is determined
3 by the following relationship:

$$4 \quad R_f = V_w/V_c = 1 + [(1-n)/n]\rho_s K_d \quad (6.6)$$

5 where: n is the aquifer porosity, assumed to be 30% for sediments in the Muscoy Plume OU (Domenico
6 and Schwartz 1990); ρ_s is the sediment particle density, assumed to be 2.65 g/cm³ for sediments in the
7 Muscoy Plume OU (Domenico and Schwartz 1990); and K_d is the adsorption, or distribution coefficient
8 discussed above. The calculated R_f for PCE is 2.75 and 1.92 for TCE. These values are within the
9 expected range of 1 to 10 for most homogenous sand and gravel aquifers that are low in solid organic
10 matter (Mackay et al. 1985), and suggest that the center of mass of the PCE and TCE plume would be
11 expected to move at about 40-50% of the average velocity of the groundwater.

12 The retardation model defined by equation (6.6) is somewhat simplified. It is possible that the extent of
13 retardation could vary considerably in space and time due to factors (e.g., contaminant characteristics,
14 presence of other contaminants, aquifer properties etc.) that influence the interaction of a contaminant and
15 the aquifer solids (Mackay et al. 1985).

16 It should be noted that contaminant migration cannot be accurately estimated using only the average
17 groundwater velocity and retardation factor. The dispersive characteristics (e.g., frictional forces,
18 varying pore sizes, path length, velocity gradients) of the aquifer can result in substantial contaminant
19 spreading and dilution. Consequently, the spreading plume front can arrive at a given location well in
20 advance of the time estimated solely on the basis of the average flow velocity and retardation factor
21 (Mackay et al. 1985).

22 **Biotransformation**

23 Many groundwater contaminants are biologically reactive. Numerous studies have shown that certain
24 aquifers can harbor appreciable populations of metabolically active microorganisms capable of degrading
25 organic contaminants. Depending upon the groundwater or aquifer environment, biotransformation of
26 contaminants can result in the formation of products that are either less harmful or potentially more toxic
27 and mobile. These reactions can occur in the aquifer or can be induced in-situ in surface treatment
28 systems supplying required nutrients, substrates, and electron donors/acceptors.

29 As previously discussed, aerobic conditions are expected to predominate in this aquifer, although some
30 minor, localized areas of anaerobic conditions could exist. There are no known data showing any sources
31 of hydrocarbons which would provide a primary substrate for biological activity. Natural sources of
32 petroleum hydrocarbons, however, may be present in the groundwater basin.

33 As discussed in Subsection 6.1.2, PCE and TCE can be co-metabolically transformed by methanotrophs,
34 a physiological group of bacteria (e.g., Methylomonas sp., Methylobacter sp., Methylosinus
35 trichosporium, Methylococcus capsulatus) that use (i.e., metabolize) methane as their primary growth
36 substrate, or source of carbon and energy. The reductive transformation under anaerobic conditions
37 favors PCE and the more chlorinated ethylenes, while oxidative transformation under aerobic conditions
38 seems to favor the less chlorinated, short-chained ethylenes.

1 A number of recent investigations have provided evidence of TCE oxidation by pure cultures of
2 methanotrophs capable of using both methanol and methane as growth substrates. Another study found
3 a bacterial strain capable of degrading TCE that could use either phenol or toluene as a growth substrate,
4 while another strain growing on citrate was reported to degrade PCE as well as TCE (McCarty 1988).
5 Although these studies indicate bacteria other than methanotrophs are capable of co-metabolizing
6 chlorinated alkenes, the methane-oxidizing bacteria are the most common and widely studied of the
7 subsurface bacterial groups.

8 The primary substrate, methane, is required to supply the major energy requirements for bacterial growth
9 and/or for the activation of enzymes necessary for the biotransformation of the contaminants. Although
10 the transformation of TCE will continue for some time in the absence of methane, it will eventually stop
11 if methane is absent for a prolonged period of time. Methane competitively inhibits the oxidation of TCE
12 since both substrates compete for the same metabolic enzyme, methane monooxygenase (MMO). The
13 proposed transformation mechanism, catalyzed by MMO, is believed to be an epoxidation reaction
14 resulting in the intracellular formation of TCE epoxide which is then extruded from the bacterial cell into
15 the aqueous environment where it undergoes spontaneous abiotic degradation to carbon monoxide,
16 formate, dichloroacetate and glyoxylate (Henry and Grbic-Galic 1991). The relative rate of oxidation
17 appears to increase for compounds containing fewer chlorine atoms per carbon. Highly chlorinated
18 compounds, such as PCE, do not appear to undergo oxidation while less chlorinated compounds such as
19 vinyl chloride are transformed very rapidly. The order of oxidation of a mixture of chlorinated ethylenes,
20 would proceed very rapidly for vinyl chloride; slower for cis-and trans-1,2-dichloroethene and 1,1-DCA;
21 even slower for TCE; and not at all for PCE (McCarty 1988).

22 Transformation under anaerobic conditions, as discussed in Subsection 6.1.2, consists of sequential
23 reductive dehalogenation reactions. The transformation or biodegradation of PCE [PCE to TCE; TCE
24 to the 1,2-DCE isomers (primarily the cis isomer); cis-1,2-DCE to vinyl chloride; and finally, to ethylene
25 and ethane] has been demonstrated by several investigators (McCarty 1988). Unlike oxidation, the
26 relative rate of reductive transformation increases for those compounds containing more chlorine atoms
27 per carbon.

28 A multitude of factors affect transformation rates and most are not well understood. The factors include
29 such things as the number and species of microorganisms present, water temperature, pH, interactions
30 between primary and secondary substrates (i.e., methane and PCE and/or TCE), sequential
31 decomposition, sorption, volatilization, microbial toxicity, and transformation reaction stoichiometry
32 (McCarty 1988). In some cases, native microorganisms are incapable of transforming a particular
33 contaminant or need an extended period of acclimation (Mackay et al. 1985). Although it has been fairly
34 well documented that acclimated microorganisms, in numbers necessary to achieve substantial
35 transformation rates, are present in shallow (<18 feet), unconfined aquifers; their presence in deep
36 aquifers has not been fully confirmed.

37 The data generated during this RI fail to provide conclusive evidence that biological transformation may
38 be occurring and attenuating groundwater concentrations, or otherwise affecting the fate and transport of
39 TCE or PCE in the Muscoy groundwater plume. The products of the reductive dehalogenation of PCE
40 (i.e., TCE and cis-1,2-DCE) were present in some groundwater samples collected from five municipal
41 supply wells (see Table 5-1 [Muni-104, -105, -106, -107, and -109]). This suggests the possibility of
42 some subsurface biological transformation under localized anaerobic conditions. However, the absence
43 of primary substrates, metabolic by-products, or geochemical indications of anaerobic conditions in the

1 Muscoy Plume OU investigation area seem to indicate otherwise. The formation of vinyl chloride,
2 ethylene, and ethane could accompany the reductive dehalogenation of PCE or TCE. However, the
3 absence of vinyl chloride, ethylene, and ethane in groundwater does not preclude the possibility that they
4 may be undergoing oxidative dehalogenation, with the formation of potential intermediates and end
5 products that are not on the Target Compound List nor included among the analytes of this RI.

6 **6.3.2 Intermedia Transport Pathways**

7 This subsection provides a discussion of the transport of contaminants from the groundwater, or existing
8 contaminant source, to potential points of human or ecological exposure. The potential pathways include
9 volatilization or flux of contaminants from the groundwater to the atmosphere (atmospheric pathway);
10 contaminated water exchange between groundwater and surface water (surface water pathway); and
11 exchange between groundwater and biota (biotic pathway).

12 **Atmospheric Pathway**

13 Because air sampling, soil gas and soil sampling from the vadose zone underlying the Muscoy Plume OU
14 were not performed as part of the investigative activities, evaluation of the transport or release of gaseous
15 or airborne particulate contaminants from the vadose and saturated zones beneath the site could not be
16 directly determined. However, based on the assumption that the source of contamination in the Muscoy
17 plume is significantly upgradient of the Muscoy Plume OU boundary and given the results of the
18 Newmark RI (where source and hydrogeologic conditions may be similar to the Muscoy plume), release
19 of airborne particulate contaminants from the vadose and/or saturated zones underlying the Muscoy Plume
20 OU could be considered an insignificant intermedia transport pathway. The results of the Newmark RI
21 are summarized below.

22 The analysis of soil cores (see Appendices C and F of the Newmark OU RI/FS Report), collected at
23 different depths during the installation of monitoring wells (MW02 through MW08) in the suspected
24 source area and the results of a soil gas survey conducted in the area of the former disposal trench (see
25 Appendix L of the Newmark OU RI/FS Report) showed no VOCs present in the vadose zone at
26 detectable levels. Consequently, emissions of contaminated fugitive dusts (i.e., contaminated soil
27 particles) or volatile releases of sorbed contaminants from the unsaturated soils underlying the formerly
28 suspected source area near the "Cat Pit" are not considered a potential human or ecological exposure
29 pathway.

30 Indoor air samples were collected from three residences and one control residence (see Appendix K of
31 the Newmark RI/FS Report) near the Cat Pit (URS 1993). PCE was present in all source area residences
32 at a range of concentrations from 0.56 parts per billion by volume (ppbv) to 0.98 ppbv. TCE was
33 present at two of the source area residences at concentrations of 0.32 ppbv to 0.47 ppbv, but was not
34 detected above the method detection limit (MDL) of 0.20 ppbv, at the third residence. Both PCE and
35 TCE were detected at concentrations of 0.35 ppbv and 0.25 ppbv, respectively in the indoor air of the
36 control residence located immediately outside the suspected source area. The PCE and TCE
37 concentrations present in the indoor air of these residences were within a range of background levels
38 determined during joint EPA and California Air Resources Board (CARB) surveys (Total Exposure
39 Assessment Methodology or TEAM studies) conducted in the southern California area in 1984 and 1987.
40 The 1987 TEAM study data found 24-hour average PCE concentrations in the indoor air of Los Angeles
41 area residences ranged from 0.21 ppbv to 1.00 ppbv, with a maximum 12-hour concentration of 7.9

1 ppbv. The 24-hour average TCE concentrations ranged from 0.15 ppbv to 0.82 ppbv, with a maximum
2 12-hour concentration of 5.23 ppbv.

3 **Surface Water Pathway**

4 The groundwater in the Muscoy Plume OU originates from surface water runoff in the San Bernardino
5 Mountains. Surface water flows into the study area across the San Andreas Fault through the outlets of
6 Devil Canyon, Badger Canyon, and Waterman Canyon (Hardt and Hutchinson 1980). Once surface water
7 has passed the base of the San Bernardino Mountains, it flows into percolation basins located downslope
8 of each canyon outlet. Most of the surface water percolates through the alluvial deposits into the
9 groundwater as enhanced groundwater recharge. Data gathered from San Bernardino Valley gauging
10 stations measuring surface-water inflow and outflow support the conclusion that, barring periods of
11 infrequent flooding, most of the surface water entering the valley seeps into the aquifer (Hardt and
12 Hutchinson 1980). Danskin and Freckleton (1989) estimate that 90% of the surface water entering the
13 valley seeps into the groundwater.

14 The streams and rivers act as natural recharge or discharge points for groundwater and surface water
15 depending on the position of the water table relative to the water level of the streams. Streams filled for
16 short durations during wet winter and spring months percolate surface water downward to the
17 groundwater. During most summer months, the streams are dry and groundwater that flows towards the
18 streams within ten feet of the ground surface is rapidly lost to the atmosphere through evapotranspiration
19 (Hardt and Hutchinson 1980). These study area recharge/discharge points are all located upgradient of
20 the Muscoy plume. Consequently, there are no known groundwater to surface water discharge points
21 located within the area of the Muscoy Plume OU.

22 **Biotic Pathway**

23 As discussed above, there is no current evidence of PCE or TCE contamination present in surface water
24 capable of supporting biotic populations (e.g., edible crops, livestock, game populations, terrestrial and
25 aquatic animals) that could serve as potential pathways for human or ecological exposure. Evidence of
26 PCE or TCE contamination in soil and air (capable of supporting biotic population) could not be
27 confirmed because sampling of these media were not performed. However, as previously discussed,
28 based on results of the Newmark OU RI and assuming similar site conditions exist in the Muscoy Plume
29 OU, potential contamination in the soil and air at the Muscoy Plume OU could be considered
30 insignificant. On this basis, only those biotic populations supported by groundwater pumped from the
31 contaminated groundwater in the Muscoy plume would constitute potential pathways for human or
32 ecological exposure.

33 Based on well pumpage data for the model area obtained from the Western Watermaster (Western
34 Municipal Water District 1986) and the area water agencies (see Subsection 6.4.2), discharge from the
35 groundwater system is limited to municipal supply wells, private water companies, and mutual water
36 companies that supply domestic drinking water within the model area. These wells constitute the public
37 water system and as such are subject to the requirements of the Safe Drinking Water Act and drinking

1 water MCLs³, including MCLs for PCE and TCE. These public water system wells currently pumping
2 contaminated groundwater (i.e., Muscoy plume groundwater) are treating or otherwise ensuring that the
3 groundwater meets both MCLs and the state recommended Action Levels. Other than the public water
4 system wells, there are no wells in the model area known to be actively pumping at this time.

5 Based on the information gathered during this RI, there is no known use of any untreated groundwater
6 from the Muscoy plume to support biotic populations. Consequently, biotic populations within the model
7 area are not considered potential human or ecological pathways for exposure to TCE or PCE.

8 **6.4 REVIEW OF PROJECT FLOW MODEL**

9 The groundwater flow model used during the Muscoy Plume OU RI/FS is the same model developed for
10 the Newmark OU RI/FS. The model was modified and recalibrated during the scoping phase of the
11 Muscoy OU RI/FS (URS 1993b).

12 The primary purpose of the project flow model was to evaluate or screen groundwater extraction and
13 injection alternatives. For the purposes of this Section, the model serves as the basis for estimating
14 contaminant migration. Development of the project flow model consisted of several processes:

- 15 ■ Development of the conceptual model;
- 16 ■ Definition of the model area;
- 17 ■ Preparation of the input data;
- 18 ■ Definition of the grid system; and
- 19 ■ Calibration of the steady-state and transient-state flow models.

20 The MODFLOW (McDonald and Harbaugh 1988) groundwater flow program was used to simulate the
21 groundwater flow for the model area. MODFLOW is a groundwater flow program capable only of
22 simulating the advection processes that take place in the groundwater system. It cannot simulate
23 contaminant transport. MODFLOW can therefore only be used to simulate the direction and, to a limited
24 extent, the rate of advective transport of dissolved TCE and PCE. PATH3D[®] (Zheng 1991) and
25 SURFER[®] (Golden Software, Inc. 1993) were used as post-processors for the MODFLOW output data.
26 PATH3D[®], a groundwater path and travel-time program, utilized the input data and unformatted head
27 files of MODFLOW simulations to:

- 28 ■ Create contours of the calculated heads;
- 29 ■ Simulate the pathlines of imaginary particles placed in the Muscoy Plume OU; and
- 30 ■ Delineate capture-zones for each extraction scenario.

31 ³ MCLs are the federally-enforceable limits for contaminants in drinking water established under the National Primary
32 Drinking Water Regulations. The MCLs are set by the Office of Drinking Water under the authority of the Safe Drinking Water
33 Act. The state may opt for Primacy Status for drinking water. States which have Primacy Status from EPA must adopt standards
34 which are at least as stringent as federal standards.

1 SURFER® (Golden Software, Inc. 1993) is a graphics program, which utilizes the head contour files
2 created by PATH3D® to produce plots displaying the head contours, particle pathlines, and the overall
3 response to groundwater extraction.

4 There were two stages in the development and calibration of the project flow model from the
5 MODFLOW program, the steady-state flow model and the transient-state flow model. Transient-state
6 groundwater movement or storage in an aquifer system reflects a change in storage due to the differences
7 in the input and output; in steady-state conditions, however, the change in storage is equal to zero since
8 the input is always equal to the output.

9 The steady-state flow model was simulated and calibrated for the time period between January 1982 to
10 January 1986. The input data and boundary conditions are described in Sections 1.5 and 2.3 of Appendix
11 J of the Newmark OU RI/FS Report (URS 1993c). The transient-state flow model was simulated and
12 calibrated for the time period between January 1986 to December 1990. The input data and boundary
13 conditions resulting from the calibration of the steady-state flow model were used as the initial conditions
14 for the transient-state flow model. Some of the input data and boundary conditions (e.g., transmissivities,
15 recharge values) were refined in order to calibrate the transient-state flow model. The calibrated
16 transient-state flow model then became the project flow model which was used for simulation of the
17 extraction scenarios for the Newmark OU RI/FS. The measured recharge, streamflow, pumpage from
18 existing municipal water supply wells, and head values for the January 1986 through December 1990
19 period were used in all extraction scenario simulations.

20 As stated at the beginning of this subsection, the calibrated flow model resulting from the Newmark OU
21 RI/FS was further modified and recalibrated during the scoping phase of the former Muscoy OU work
22 assignment (Appendix 5). Model run 55A was used as the starting point for predictive simulations as
23 part of this WA (Muscoy Plume OU RI/FS). Measured recharge, streamflow, pumpage, and head values
24 for the January 1986 through December 1990 period were also used in all scenarios for the Muscoy
25 Plume OU extraction simulations.

26 Subsection 6.4.1 includes a summary of the modifications and calibrations conducted on the project flow
27 model during the scoping phase of the RI/FS. That effort was documented in a technical modeling
28 memorandum dated November 1993 and attached to this RI/FS Report as Appendix 5. Specific details
29 of input parameters and initial conditions used in the project flow model are contained in Appendix J of
30 the Newmark OU RI/FS Report and are not included here for brevity. The reader is referred to
31 Appendices 5 and 6 for additional details about the project flow model development. As discussed in
32 Subsection 4.2, in the area of the Muscoy Plume OU, the flow model consisted of two layers: an
33 unconfined layer (layer 1) and a confined layer (layer 2).

34 **6.4.1 Summary of Model Changes**

35 The primary objectives of the scoping phase modeling effort were to identify areas of high uncertainty
36 and critical data gaps in the Muscoy Plume OU. The overall purpose of the model study was to evaluate
37 the feasibility of a groundwater extraction system; and, if feasible, provide a technical basis for selection
38 of an extraction scenario to contain groundwater contamination in the Muscoy Plume OU.

39 A preliminary series of model runs (24) was conducted to identify data gaps and to evaluate what impacts
40 that these data gaps may have on the model. These attempts to further calibrate the model, concentrating

1 on the Muscoy Plume OU, were conducted through an iterative process. Calibration was sufficiently
2 successful for use during the FS and further model improvement may be implemented as focused
3 information is gathered during the Muscoy Plume Remedial Design and the Source OU RI/FS (Appendix
4 5).

5 The project flow model predicts reasonable responses to pumping, given the current input data set and
6 calibration state. Model runs 51B, 51C, and 51D were conducted to simulate known pumping rates from
7 the Baseline Feeder wellfield. This wellfield consists of municipal supply wells near 9th Street and Mt.
8 Vernon Avenue and 9th and Perris Streets. The pumping rates were obtained from the San Bernardino
9 Valley Municipal Water District report, "Baseline Feeder Wells, Ninth and Perris Street, Results of
10 Drilling, Testing, and Recommended Pump Design," dated May 1990. If the testing results are assumed
11 to represent accurate, long-term head drawdowns, then model capture predictions should be conservative
12 because the simulated drawdowns are less than the drawdowns calculated in the test analysis. Uncertainty
13 still exists, however, since the test analysis was based on a relatively short pumping period (24 hours)
14 and drawdowns observed after this pumping period were linearly projected up to one year. If the linear
15 basis for the pumping test projections is not valid, then the project flow model simulations may be less
16 conservative.

17
18 Preliminary model calibration after run 55A was considered complete. The project flow model was
19 considered sufficiently calibrated to predict pumping scenarios during the FS; however, a greater level
20 of certainty in the model is needed before it should be applied to RD. The project flow model will be
21 updated as data from the Source OU RI/FS are collected. Model run 55A was considered the best
22 calibration run and was used as a base for subsequent simulations during Muscoy Plume OU predictive
23 modeling efforts.

24 **6.4.2 Flow Model Results of Existing Conditions**

25 The transient-state flow model, calibrated for the time period from January 1986 through December 1990,
26 was run for a simulated time period of 35 years. Model runs 56A, 56B, and 57A were conducted to
27 evaluate the ability of existing wellfields to control contaminant migration in the Muscoy Plume OU. The
28 predictive run known as the "no action" scenario (Run 57A) is discussed in this Subsection.

29 The input data (including quarterly well pumpage) and boundary conditions used in the calibration of the
30 transient-state flow model (from run 55A) were applied to Run 57A for the first 5 years of the simulation.
31 These conditions were repeated in 5-year cycles for 30 years.

32 The existing municipal supply wells, which were pumping between January 1986 through December
33 1990, were used through the 35-year simulation. Table 12-1 lists the locations of the existing municipal
34 supply wells and their pumping rates.

35 To create imaginary particles, three sets of imaginary particles (total of 54) were used in PATH3D[®]. Set
36 1 contained seventeen imaginary particles that were placed near the northern portion of the Muscoy Plume
37 OU investigation area along a northeast-southwest transect. Set 2 contained eighteen imaginary particles
38 that were placed approximately half-way between the northern portion of the Muscoy Plume OU and the
39 19th Street wellfield along a northeast-southwest transect. Set 3 contained 19 particles that were placed
40 just south of the 19th Street wellfield along a northeast-southwest transect.

1 The pathline of an imaginary particle produced by PATH3D[®] represents movement of groundwater in
2 the aquifer with time. Since the contaminants (TCE and PCE) move with the groundwater, the imaginary
3 particle pathline also represents the movement of contaminants in the aquifer with time. The pathlines
4 of the 54 imaginary particles placed, as described before, in the plume area represent the movement of
5 contaminants in the Muscoy Plume OU. Effectiveness of an extraction scenario was evaluated based on
6 the capture of imaginary particles by the extraction wells.

7 SURFER[®] was used to produce plots of the head contours and pathlines created during the application
8 of PATH3D[®].

9 Results and Summary

10 The head contour plots for layers 1 and 2 and the PATH3D[®] output file were analyzed for the no action
11 scenario (extraction scenario no. 1). Figures 6-2 and 6-3 display the head contour plots for the end of
12 the 35-year simulation for layers 1 and 2, respectively. These figures also display the imaginary particles
13 and their pathlines.

14 Extraction scenario no. 1 consisted of extraction from the 19th Street and Baseline Feeder wellfields.
15 The extraction from the 19th Street wellfield was as follows:

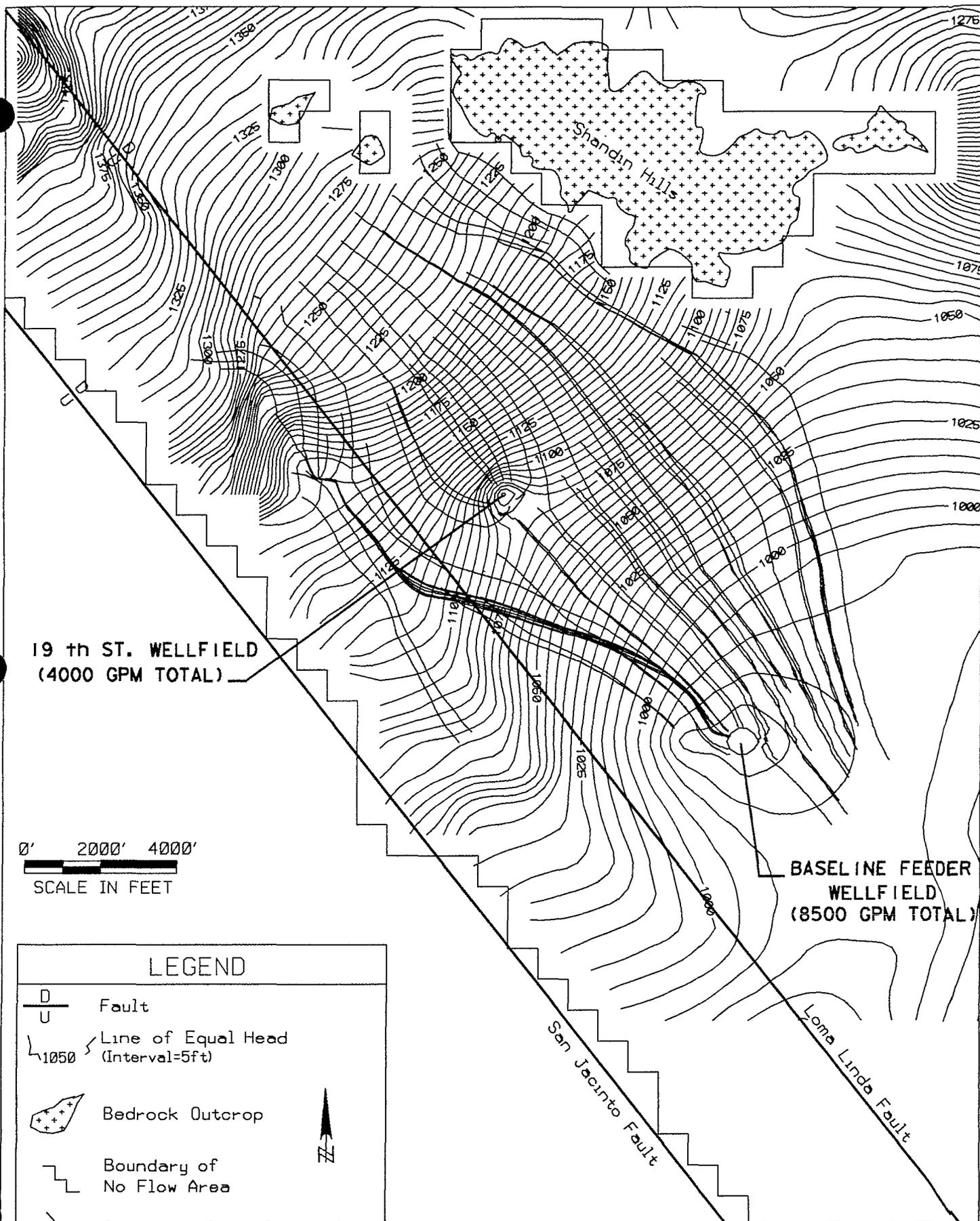
- 16 ■ For the first 5-year period between January 1986 to December 1990, actual pumping
17 rates were used; and
- 18 ■ For the next 30-year period, constant daily pumping rate of 2000 gpm from each of the
19 19th Street #1 and #2 wells was considered.

20 The 30-year period pumping rates represented an increased pumping compared to the normal pumping
21 rate. Based on a report by Geoscience 1990, the following extraction was considered for the Baseline
22 Feeder wellfield:

- 23 ■ No pumping for the first 5-year period; and
- 24 ■ For the next 30-year period, constant daily pumping rate of 4000 gpm and 4500 gpm
25 from Perris Street and 9th Street wells, respectively.

26 Figures 6-2 and 6-3 show the head contours and pathlines of imaginary particles for layers 1 and 2,
27 respectively. A few imaginary particles were captured by the existing wellfields. A few imaginary
28 particles near the San Jacinto Fault and many particles in the southern portion of the Muscoy Plume OU
29 were not captured.

30 Based on the results of extraction scenario 1, the 19th Street wellfield (municipal supply wells no. 1 and
31 no. 2 near 19th and Flores Streets) appeared to capture most of the nearby imaginary particles. However,
32 most of the particles near the Baseline Feeder wellfield were not captured.



19th ST. WELLFIELD
(4000 GPM TOTAL)

BASELINE FEEDER
WELLFIELD
(8500 GPM TOTAL)

0' 2000' 4000'
SCALE IN FEET

LEGEND	
	Fault
	Line of Equal Head (Interval=5ft)
	Bedrock Outcrop
	Boundary of No Flow Area
	Imaginary Particle Pathline

FIGURE 6-2
HEAD CONTOUR AND PATHLINE PLOT
FOR EXTRACTION SCENARIO NO. 1
LAYER 1 (UPPER AQUIFER)

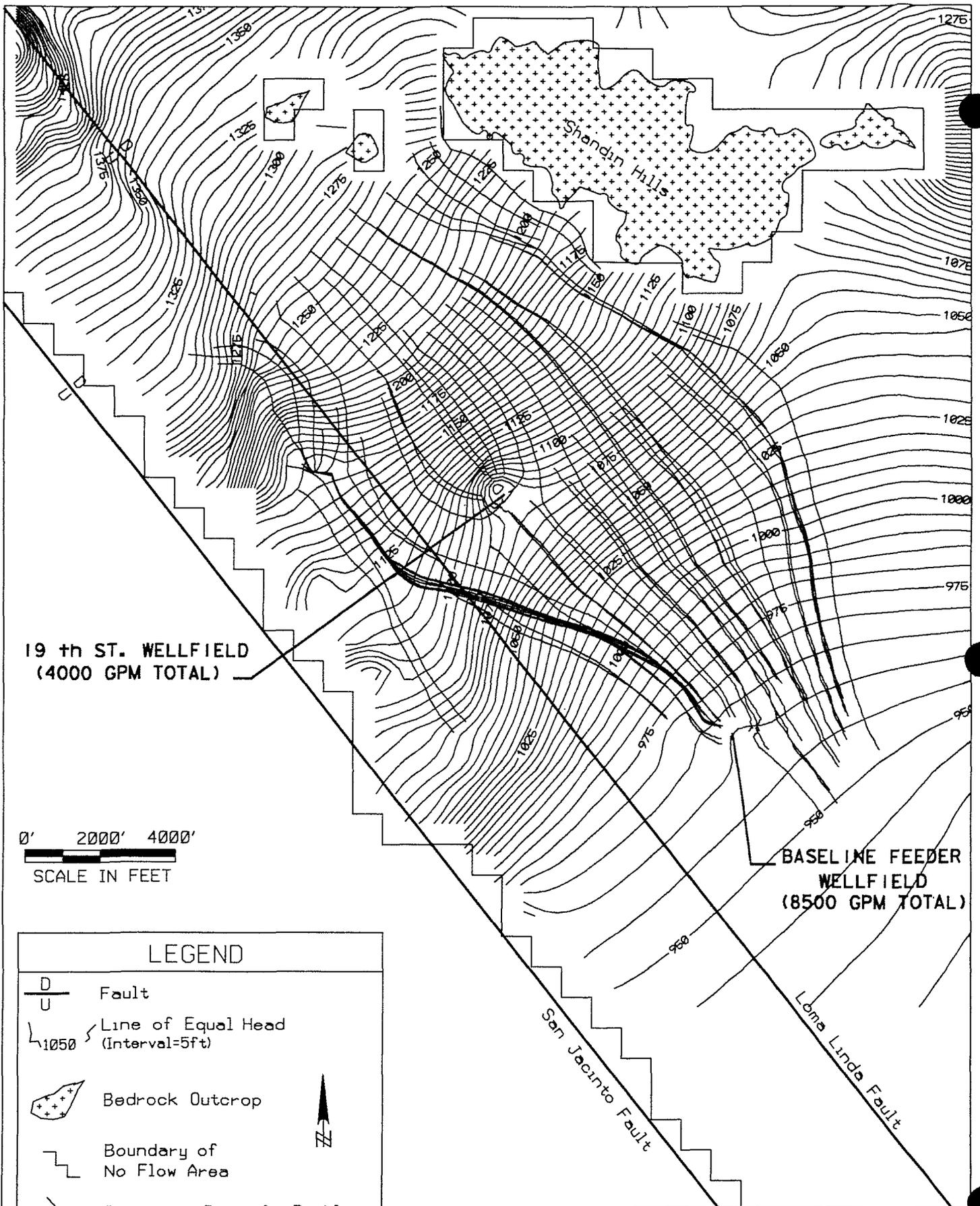


FIGURE 6-3
 HEAD CONTOUR AND PATHLINE PLOT
 FOR EXTRACTION SCENARIO NO. 1
 LAYER 2 (LOWER AQUIFER)

1 At the present time, the groundwater contamination is approximately 10,000 feet (1.9 miles) long
2 measured from the northern portion of the Muscoy Plume OU investigation area (Figure 5-1). It is
3 approximately 8,000 feet (1.5 miles) wide at its widest point adjacent to the southwest edge of Shandin
4 Hills. Based on the positions of the imaginary particles and assuming no retardation of the contaminant
5 velocities, the extent of the contamination will be approximately 18,000 feet (3.4 miles) long from the
6 northern portion of the Muscoy Plume OU (Figure 5-1) after 35 years of migration. It will be
7 approximately 8,000 feet (1.5 miles) wide at its widest point adjacent to the southwest edge of Shandin
8 Hills. Therefore, according to the results of the project flow model as shown in Figures 6-2 and 6-3,
9 the contaminant plume could be expected to migrate approximately 8,000 feet (1.5 miles) downgradient
10 in 35 years.

11 The following two groups of existing water-supply wells within the Muscoy Plume OU captured
12 imaginary particles during the simulation and, therefore, may have an influence on the plume.

13 1) The 19th Street wellfield captured particles on the western side of the plume.

14 2) The Baseline Feeder wellfield also captured particles near the leading edge of the plume.

15 The 19th Street wellfield contained very small amounts of contamination. The Baseline Feeder wellfield
16 reportedly has had no detectable contamination to date.

17 Based on the present configuration of the model and pumping values from 1987-1991 data, the most
18 conservative model projections estimate that the Muscoy plume edge will reach the Baseline Feeder
19 wellfield area circa 2003. This estimate does not consider contaminant retardation nor, on the other
20 hand, that actual groundwater velocities may differ from the average groundwater velocity here.

21 An average groundwater velocity was calculated for the Muscoy Plume OU investigation area. The
22 groundwater velocity was represented by the velocities of the imaginary particles that were placed in the
23 OU. The velocity of an imaginary particle equaled the travel-distance divided by the travel-time.
24 Imaginary particles that were not affected by the boundary conditions of the project flow model and
25 plotting limitations of SURFER[®] and PATH3D[®] were used to calculate an average groundwater velocity
26 for the investigation area. Table-6-2 presents the calculated average groundwater velocity for the Muscoy
27 Plume OU investigation area.

28 The estimated average groundwater velocity for the Muscoy Plume OU investigation area is a simple
29 arithmetic average. The estimate of average groundwater velocity using particle travel times was
30 considered a good estimate for the following reasons:

31 ■ The project flow model is a computer model and, therefore, groundwater velocities
32 provided by the project flow model are estimates or averages over a much larger distance
33 than estimates about a single well using pumping test derived data;

Table 6-2

AVERAGE GROUNDWATER VELOCITIES FOR MUSCOY PLUME OU

Velocity (ft/day)	Velocity (ft/yr)
Average Groundwater Velocity	
1.4	500

1
2
3
4
5
6
7

- It is not known how the groundwater velocity estimated for the Muscoy Plume OU investigation area compares with actual field conditions since they have not been measured. The calculated velocity could not be verified but represents the best possible estimate over the length of the investigation area. There are no reasonable methods for directly measuring groundwater velocity in this basin since tracer studies are neither feasible nor warranted as this time. Indirect measurements are good alternatives but result in some uncertainty.