# 5.0 CONTAMINANT FATE AND TRANSPORT

## 5.1 INTRODUCTION

This chapter describes the potential migration pathways and mechanisms for transport of chemical substances found in surface and subsurface soils, surface water, and groundwater at WBG. Computer-based contaminant fate and transport modeling analyses were performed to predict rate of contaminant migration in the identified primary media and project likely future contaminant concentrations at receptor locations through these media. The ultimate objectives of these analyses are to evaluate potential future impacts to human health and the environment and to provide a basis for evaluating the effectiveness of the proposed remedial alternative in the FS.

Fate and transport modeling was used to simulate contaminant distribution in the WBG surface water system comprising overland flow from drainage areas to the drainage ditches and tributaries. Surface water and sediment transport modeling was performed to predict future contaminant concentrations in the tributaries flowing west to east across WBG and entering Sand Creek. Surface water concentrations in Sand Creek due to contaminant loading from WBG were also simulated. Neither groundwater nor air quality modeling was performed for this RI as these media were not identified as primary pathways of contaminant transport at WBG.

A summary of the principles of contaminant fate and transport is presented in this chapter along with the results of the surface water and sediment transport modeling activities. Section 5.2 presents a conceptual model for potential contaminant migration pathways at WBG that considers site topography, hydrogeology, contaminant sources and distribution in various environmental media, and release mechanisms through transport media. Section 5.3 presents the persistence of contaminants in the environment and the physical and chemical properties of the SRCs (including metals, organic compounds, and explosives found at the WBG). Section 5.4 presents the numerical modeling of contaminant transport via surface water (the primary transport medium at WBG). The summary and conclusions of the fate and transport analysis are presented in Section 5.5.

## 5.2 CONCEPTUAL SITE MODEL

The CSM is a statement of known site conditions that serves as the framework for quantitative modeling. Site conditions described by the CSM include waste source information, the surrounding geologic and hydrologic conditions, and the SRCs and their current spatial distribution. This information is combined to identify chemical migration pathways at WBG. The predictive function of the CSM, of primary importance to contaminant fate and transport analysis, relies on known information and informed assumptions about the site. The better the information and the greater the accuracy of the assumptions, the more accurately the CSM describes the AOC and, therefore, the more reliable the predictions.

The details of the CSM have been presented in Sections 3.0 and 4.0. A summary of the salient model elements follows.

### 5.2.1 Contaminant Sources

Based on historical records and the findings of sampling and analysis at WBG, the following contaminant sources have been identified:

- individual burning pads and roadside ditches that were used periodically to destroy explosives and other materials by burning;
- surrounding surface soils that appear to act as a source of surface runoff;
- surrounding subsurface soils, specifically near the contaminated pads; and
- the crushed slag that was used throughout WBG for roads, pads, and driveways.

### 5.2.2 Hydrogeology

A complete description of the site geology and hydrology is provided in Section 3.0 and can be summarized as follows:

- The topography consists of gently undulating slopes and level areas decreasing from west to east across WBG.
- In general, WBG is covered by low-permeability soils and glacial sediments except where the natural material have been either eroded, removed, reworked, or covered during RVAAP operations.
- Most surface water flows from west to east and north to southeast across WBG in three small streams, eventually entering Sand Creek. Mack’s Pond, located in the southwest quadrant of WBG, is fed by a surface water channel draining the western portion of WBG. The extreme northwest corner of WBG (Pads 58-61) drains northeastward toward the pistol range.
- Groundwater is present in the sandy interbeds found in glacial materials that occur within approximately 7.6 m (25 ft) of the ground surface. The glacial material present at WBG is presumed to be tens of feet thick. The water-bearing units behave as unconfined systems. Based on topography of the site and potentiometric surface data, groundwater is presumed to flow from west to east and southeast with an average velocity varying between $2.12 \times 10^{-2}$ to $5.65 \times 10^{-4}$ cm/s (see Chapter 3.0).

### 5.2.3 Water Balance

The potential for contaminant transport begins with precipitation. The actual amount of rainwater available for flow is highly variable and dependent upon soil type and climatic conditions. A water balance calculation can be used as a tool to quantitatively account for all the components of the hydrologic cycle at WBG. The components of a simple steady-state water balance model include precipitation (P), evapotranspiration (ET), surface runoff (Sr), and groundwater recharge or percolation (Gr) and is defined as follows:

$$P = ET + Sr + Gr \,,$$

or

$$\text{Rainwater available for flow} = \quad Sr + Gr = P - ET \,.$$

A relatively moderate amount of runoff occurs from the site. Of the total annual precipitation of 36.8 inches, runoff amounts estimated by the SWMM model are 13.6 inches. Contribution of the site to runoff is 37 percent of annual precipitation. It is expected that there will be loss of runoff water in the form of evaporation. The remaining water after runoff is infiltration, which includes loss to atmosphere by evapotranspiration. The WBG water balance components are summarized in **Figure 5-1**.

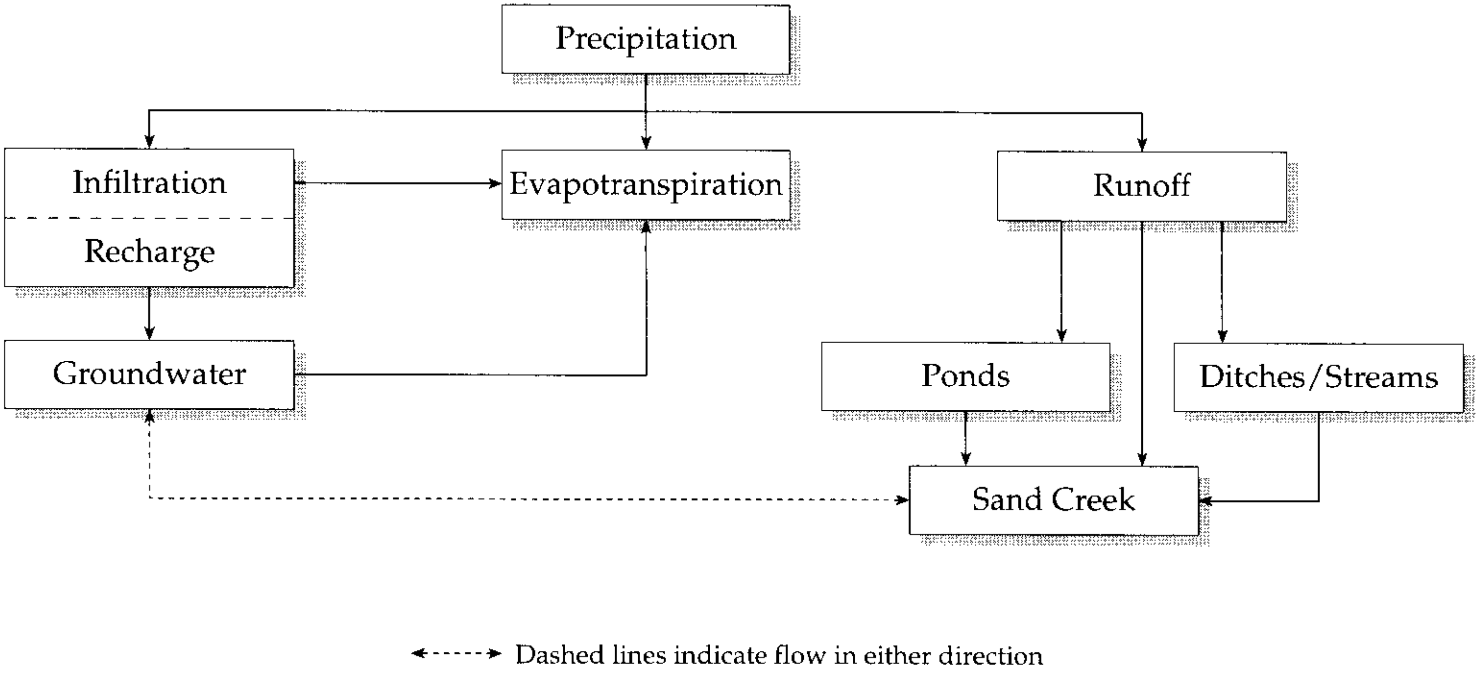

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Figure 5-1. Water Balance Components at WBG

### 5.2.4 Site-Related Chemicals

The SRCs for WBG are identified in Section 4.1.4. The SRCs represent a comparison of detected maximum concentrations of the chemicals against their respective background criteria. The SRCs are summarized in **Tables 5-1 through 5-3**.

**Table 5-1. List of Distribution Coefficients ($K_d$ s) for Inorganic SRCs at WBG**

| SRCs | Range of $K_d^a$ in loam type soil (L/kg) | Range of $R_d^e$ |
|---|---|---|
| Aluminum | $1500^b$ | 7500 |
| Antimony | 150 (100 to 10000) | 751 (500 to 50,000) |
| Arsenic | 2.0E+02[b] | 1000 |
| Barium | $50^b$ | 251 |
| Beryllium | 800 | 401 |
| Cadmium | 40 (7 to 962) | 201 (36 to 4810) |
| Chromium | 30 (2.2 to 1000) | 151 (12 to 5000) |
| Cobalt | 1,300 (100 to 9700) | 6,500 (501 to 48,500) |
| Copper | 3.5E+01[b] | 176 |
| Cyanide | $9.9^c$ | 50.5 |
| Lead | 16000 ($100^c$ to 59000) | 80,000 (501 to 295,000) |
| Manganese | 750 (40 to 7.7E+04) | 3750 (201 to 385,000) |
| Mercury | 200 (10 to 10,000)[d] | 1000 (51 to 50,000) |
| Nickel | 300 | 1500 |
| Selenium | 150 | 751 |
| Silver | 120 (28 to 3.33E+02) | 601 (141 to 1666) |
| Thallium | $96^c$ to 15000 | 481 to 75,000 |
| Zinc | 1300 (3.6 to 11,000) | 6500 (19 to 55,000) |

[a] $K_d$ values taken from Sheppard and Thibault (1990) unless otherwise noted; the range is provided in parentheses if available.
[b] $K_d$ values obtained from Baes et al. (1984)
[c] Source: EPA Soil Screening Guidance Document (1996b)
[d] Source: Looney, Grant, and King (1987)
[e] $R_d$ (retardation factor) = $1 + (K_d \times \rho_b)/\phi_w$,
where:

$\rho_b$ = soil bulk dry density, (kg/L),
$\phi_w$ = soil moisture content, (unitless).

### 5.2.5 Contaminant Distribution in the Environment

Based on the findings presented in Section 4.0, the following conclusions can be made about the distribution of contaminants in WBG sources and the surrounding environmental media:

## Table 5-2. Physical and Chemical Properties of Organic SRCs at WBG[a]

| Site-Related Chemicals | Mol. Wt. | Solubility $S_w$ (mg/L) | $S_w$ @ Temp. (°C) | $K_{OW}$ (ml/ml) | Vapor Pressure (tor @ °C) | Henry's Constant ($K_h$) (atm.m$^3$/mol) | $K_h$ @ Temp. (°C) | Air Diff. Coeff. (cm$^2$/s) | $K_{OC}$ (ml/g) | Biodegradation Rate (λ) (1/day) | Log ($K_{OW}$) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| ***VOCs*** | | | | | | | | | | | |
| Acetone | 58.1 | 1.00E+06 | | 5.75E-01 | 270 @ 30 | 5.14E-07 | 25# | 0.110 s | 3.63E-01 | 2.48E-02 | -0.24 |
| Chloroform | 119.4 | 9.30E+03 | 25 | 9.33E+01 | 160 @ 20 | 3.39E-03 | 25 | 0.09 s | 5.30E+01 m | 3.85E-04 | 1.97 |
| Methylene chloride | 84.9 | 1.67E+04 | 25 | 1.78E+01 | 429 @ 25 | 3.19E-03 | 25 | 0.104 s | 1.00E+01 m | 6.19E-03 | 1.25 |
| Toluene | 92.1 | 5.15E+02 | 20 | 4.90E+02 | 28 @ 25 | 5.92E-03 | 25 | 0.087 t | 1.40E+02 m | 3.30E-03 | 2.69 |
| ***SVOCs*** | | | | | | | | | | | |
| 2-Methylnaphthalene | 142.2 | 2.60E+01 | 25 | 7.24E+03 | 10 @ 105 | 2.20E-02 | | 0.056 | 4.56E+03 | | 3.86 |
| Acenaphthene | 154.2 | 3.42E+00 | 25 | 8.32E+03 | | 2.41E-04 | 25 | 0.062 | 5.24E+03 | 1.70E-03 | 3.92 |
| Anthracene | 178.2 | 1.29E+00 | 25 | 2.82E+04 | 1.95E-4 L | 8.60E-05 | 25 | 0.042 | 2.35E+04 m | 3.77E-04 | 4.45 |
| Benzo(*a*)anthracene | 228.3 | 1.00E-02 | 24 | 4.07E+05 | 5E-9 @ 20 | 2.94E-08 | 25# | 0.051 | 3.58E+05 m | 2.55E-04 | 5.61 |
| Benzo(*a*)pyrene | 252.3 | 3.80E-03 | 25 | 9.55E+05 | 5E-9 @ 21 | 4.90E-07 | 25 | 0.043 | 9.69E+05 m | 3.27E-04 | 5.98 |
| Benzo(*b*)fluoranthene | 252.3 | 1.00E-03 | # | 3.72E+06 | 5E-7 L | 2.94E-07 | 25# | 0.044 # | 2.34E+06 | 2.84E-04 | 6.57 |
| Benzo(*g,h,i*)perylene | 276.3 | 2.60E-04 | 25 | 1.70E+07 | 1E-10 @ 20 | 1.40E-07 | | 0.042 | 1.07E+07 | 2.67E-04 | 7.23 |
| Benzo(*k*)fluoranthene | 252.3 | 3.00E-04 | # | 6.92E+06 | 1E-11 @ 20 | 2.54E-05 | 25# | 0.044 | 4.36E+06 | 8.10E-05 | 6.84 |
| Bis-(2-Ethylhexyl)phthalate | 390.6 | 1.30E+00 | 25 | 2.00E+05 | 1.2 @ 200 | 3.00E-07 | 20 | 0.032 s | 1.11E+05 m | 1.78E-03 | 5.30 |
| Carbazole | | 7.48E+00 | | | | 1.53E-08 | | | 3.39E+03 | | 3.59 |
| Chrysene | 228.3 | 6.00E-03 | 25 | 4.07E+05 | 6.3E-9 @25 | 1.05E-06 | 25 | 0.046 | 2.57E+05 | 1.73E-04 | 5.61 |
| Dibenzo(*a,h*)anthracene | 278.4 | 5.00E-04 | 25 | 9.33E+05 | 1E-10 L | 7.30E-08 | 25 | 0.042 | 1.79E+06 m | 1.84E-04 | 5.97 |
| Dibenzofuran | 168.2 | 1.00E+01 | | 1.32E+04 | | | | 0.068 | 8.31E+03 | 6.19E-03 | 4.12 |
| Di-*n*-butylphthalate | 278.4 | 4.00E+02 | 25 | 1.58E+05 | 0.1 @ 115 | 2.80E-07 | 25 | 0.042 | 1.57E+03 m | 3.01E-02 | 5.20 |
| Fluoranthene | 202.3 | 2.65E-01 | 25 | 2.14E+05 | 5E-6 L | 6.50E-06 | 25 | 0.069 s | 4.91E+04 m | 3.94E-04 | 5.33 |
| Fluorene | 166.2 | 1.90E+00 | 25 | 1.51E+04 | | 1.17E-04 | 25 | 0.055 | 9.54E+03 | 2.89E-03 | 4.18 |
| Indeno(1,2,3-*c,d*)pyrene | 276.3 | 1.40E-04 | | 4.57E+07 | 1E-10 L | 6.95E-08 | 25 | 0.044 | 2.88E+07 | 2.37E-04 | 7.66 |
| Naphthalene | 128.2 | 3.00E+01 | 25 | 2.34E+03 | 0.082 @ 25 | 4.83E-04 | 25 | 0.059 | 1.19E+03 m | 2.69E-03 | 3.37 |
| Phenanthrene | 178.2 | 8.16E-01 | 21 | 2.88E+04 | 1 @ 118 | 3.93E-05 | 25 | 0.054 | 1.82E+04 | 8.66E-04 | 4.46 |
| Pyrene | 202.3 | 1.60E-01 | 26 | 1.51E+05 | 2.5 @ 200 | 5.10E-06 | 25 | 0.051 | 6.80E+04 m | 9.12E-05 | 5.18 |

[a] Solubilities, Henry's Law Constant, and Log ($K_{ow}$) have been taken from EPA Risk Reduction Engineering Laboratory Treatability Data Base (EPA 1994a) except where otherwise indicated.
Biodegradation half-lives are taken from Hand Book of Environmental Degradation Rates (Howard et. al. 1991) except where otherwise indicated.
Air diffusion coefficients are obtained from EPA 1987, except otherwise indicated.
**[ s ]** *Source* = Shen, Schmidt, and Card 1993; **[ # ]** *Source* = EPA 1991a; m = measured $K_{oc}$ (EPA 1996b); [L] *Source* = EPA 1995.

## Table 5-3. Physical and Chemical Properties of Explosive-Related Compounds at WBG[a]

| Site-Related Chemicals | Mol. Wt. | Solubility $S_w$ (mg/L) | $S_w$ @ Temp. (°C) | $K_{OW}$ (ml/ml) | Vapor Pressure (tor @ °C) | Henry's Constant ($K_h$) (atm.m$^3$/mol) | $K_h$ @ Temp. (°C) | Air Diff. Coeff. (cm$^2$/s) | $K_{OC}$ (ml/g) | Biodegradation Rate (λ) (1/day) | Log ($K_{OW}$) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1,3,5-Trinitrobenzene | 213.1 | 3.50E+02 | | 1.58E+01 | 2E-5 C | 1.60E-08 | C | 2.49E-01 # | 9.98E+00 | NF | 1.20 |
| 1,3-Dinitrobenzene | 168.1 | 4.69E+02 | 15 | 3.09E+01 | 9E-4 C | 2.30E-07 | C | 2.80E-01 | 1.95E+01 | 1.90E-03 | 1.49 |
| 2,4,6-Trinitrotoluene | 227.1 | 1.00E+02 | 25 | 3.39E+05 | .046 @ 82 | 2.00E-07 | | 1.79E-01 # | 2.13E+05 | 1.90E-03 | 5.53 |
| 2,4-Drinitrotoluene | 182.1 | 2.70E+02 | | 1.02E+02 | .0013 @ 59 | 9.26E-08 | | 2.00E-01 | 9.55E+01 x | 1.90E-03 | 2.01 |
| 2,6-Drinitrotoluene | 182.1 | 1.82E+02 | | 7.41E+01 | 6 @ 150 | 7.47E-07 | | 2.00E-01 # | 6.92E+01 x | 1.90E-03 | 1.87 |
| 2-Nitrotoluene* | 137.1 | 4.42E+02 | 30 | 2.34E+02 | 0.1 @ 20 | 4.10E-05 | 20 | 2.30E-01 # | 1.48E+02 | NF | 2.37 |
| 3-Nitrotoluene* | 137.1 | 4.42E+02 | 30 | 2.34E+02 | 0.1 @ 20 | 4.10E-05 | 20 | 2.30E-01 # | 1.48E+02 | NF | 2.37 |
| 4-Nitrotoluene | 137.1 | 4.42E+02 | 30 | 2.34E+02 | 0.1 @ 20 | 4.10E-05 | 20 | 2.30E-01 # | 1.48E+02 | NF | 2.37 |
| HMX | 296.2 | NF | | NF | NF | NF | | NF | NF | NF | NF |
| Nitrobenzene | 123.1 | 1.90E+03 | 20 | 6.92E+01 | 0.15 @ 20 | 2.38E-05 | 25 | 7.60E-02 | 1.19E+02 m | 1.80E-03 | 1.84 |
| Nitrocellulose as N | 504.3 | NF | | NF | NF | NF | | NF | NF | NF | NF |
| Nitroglycerin | 227.1 | 1.80E+03 | | NF | 2.6E-4 @ 20 | NF | | NF | NF | NF | NF |
| RDX | 222.3 | 6.00E+01 | 23 | 7.41E+00 | NF | NF | | NF | 4.67E+00 | NF | 0.87 |
| Tetryl | 287.2 | insoluble | | NF | NF | NF | | NF | NF | NF | NF |

[a] Solubilities, Henry's Law Constant, and Log ($K_{ow}$) have been taken from EPA Risk Reduction Engineering Laboratory Treatability Data Base (EPA 1994a) except where otherwise indicated.
Biodegradation half-lives are taken from Hand Book of Environmental Degradation Rates (Howard et. al. 1991) except otherwise indicated.
Air diffusion coefficients are obtained from EPA 1987, except where otherwise indicated.
**[ s ]** *Source* = Shen, Schmidt, and Card 1993; **[ # ]** *Source* = EPA 1991a; m = measured $K_{oc}$ (EPA 1996b); [L] *Source* = EPA 1995; [C] *Source* = SCDM (EPA 1997a)
NF – Not found

- Explosives, metals, and PAHs contamination in surface soils is generally found on the pads. Concentrations of explosives are greatest at Pads #66 and 67. However, in 55 of 70 pads sampled in the Phase I RI, the residual levels of explosives were less than 1 mg/kg.

- Metals contamination appears to be distributed in the surface soil around some burning pads as well as on the pads. At some locations metal contamination has migrated off the pads. The residual levels of lead were less than 100 mg/kg. At some locations metal contamination has migrated off the pads.

- Some subsurface soils explosive contamination in the 0.6- to 1.2-m (2- to 4-ft) depth interval occurs at Pads #60, 66, and 67 (see **Figures 4-22, 4-30,** and **4-33**). Cadmium, lead, and zinc are the only metals frequently detected in subsurface soils above their background criteria. Concentrations of both explosives and metals decrease with depth.

- Several PAHs were detected in one sediment sample from the upstream end of Mack's Pond. Metals above background criteria were observed in the sediment from the eastern end of WBG.

- Acetone is the only SRC in surface water, which may be attributed to probable lab contamination.

- Migration of chemicals to groundwater is limited with minor detection of explosives and one of nine detections above background for manganese. Manganese appears to be naturally occurring as the background for manganese in groundwater is significantly high (i.e., greater than EPA Region IX criteria) in this site.

### 5.2.6 Contaminant Release Mechanisms and Migration Pathways

Based on the information presented above, the following contaminant release and migration pathways have been identified:

- Contaminants in surface soils are released to the surrounding environment through dissolution by rainwater and mass movement by soil erosion. These contaminants enter the drainage ditches/streams and are transported in surface water as either particulate-bound or dissolved chemical species. Sediment deposition, scour, suspension, and transport in surface water system may be a potential pathway for migration of contaminants.

- Water infiltrating through contaminated surface and subsurface soils may leach contaminants into the groundwater. The factors that affect leaching rate include a contaminant's solubility, $K_d$, and the amount of infiltration. Whether it is a contaminant's partitioning coefficient or solubility that controls leaching, depends on whether leaching is solubility-controlled or sorption-controlled. Insoluble compounds will precipitate out of solution in the subsurface or remain in their insoluble forms with little leaching. For the contaminants detected at WBG, generally sorption processes and the $K_d$ will have the greatest effect on leaching. Metals detected at WBG have high $K_d$s and are not expected to leach to groundwater at any significant rate. This point is supported by the fact that metals concentrations found in groundwater are below background with limited exception, despite the presence of high concentrations of metals in subsurface soils in some parts of the site. Another factor that affects whether a contaminant will reach the water table through infiltration of rainwater is the contaminant's rate of decay. Most of the organic compounds decay at characteristic rates that are described by the substance's half-life. For a given percolation rate, those contaminants with long half-lives have a greater potential for contaminating groundwater than those with shorter half-lives. However, at WBG the chemicals with longer half-lives (PAHs) have high $K_d$s, thereby limiting their migration potential to groundwater.

Release by gaseous emissions and airborne particulates is not significant at WBG. VOCs were not found in surface soils as they have already volatized over the last 40 to 50 years, therefore there is likely little to no gaseous emissions. Thus, contaminant levels in the air pathway are minor to nonexistent. Particulate matter from contaminated surface soil can become airborne as a result of wind erosion. At WBG, this process is moderated by the presence of vegetative cover throughout a large portion of the AOC.

## 5.3 PHYSICAL AND CHEMICAL PROPERTIES OF SITE-RELATED CHEMICALS

The migration of chemical constituents through various media is governed by the physical and chemical properties of the detected chemicals and the surface and subsurface media through which the chemicals are transferred. In a general way, chemical constituents and structures with similar physical and chemical characteristics will show similar patterns of transformation, transport, or attenuation in the environment. Solubility, vapor pressure data, chemical partitioning coefficients, degradation rates, and Henry's Law Constant, provide information that can be used to evaluate contaminant mobility, in the environment. Partitioning coefficients are used to assess the relative affinities of compounds for solution or solid phase adsorption. However, the synergistic effects of multiple migrating compounds and the complexity of soil/water interactions, including pH and oxidation-reduction potential (Eh), grain size, and clay mineral variability, are typically unknown.

The physical properties of the chemical constituents that were detected in the environmental media at WBG are summarized in **Tables 5-1, 5-2**, and **5-3**. The properties are used to assess the anticipated behavior of each compound under environmental conditions.

The water solubility of a compound is a measure of the saturated concentration of the compound in water at a given temperature and pressure. The tendency for a compound to be transported by groundwater is directly related to its solubility and inversely related to both its tendencies to adsorb to soil and to volatilize from water (OGE 1988). Compounds with high water solubilities tend to desorb from soils and sediments, are less likely to volatilize from water, and are susceptible to biodegradation. The water solubility of a compound varies with temperature, pH, and the presence of other dissolved constituents (including organic carbon and humic acids).

The octanol-water partition coefficient ($K_{ow}$) is a laboratory-determined ratio of the concentration of a chemical in the n-octanol phase of a two-phase system to the concentration in the water phase. The octanol-water partition coefficient can be used to estimate the tendency for a chemical to partition between environmental phases of different polarity. Compounds with log $K_{ow}$ values less than 1 are highly hydrophilic, while compounds with log $K_{ow}$ values greater than 4 will partition to soil particles (Lyman, Reehl, and Rosenblatt 1982).

The water/organic carbon partition coefficient ($K_{oc}$) is a measure of the tendency of a compound to partition between soil and water. The $K_{oc}$ is defined as the ratio of the absorbed compound per unit weight of organic carbon to the aqueous solute concentration. This coefficient can be used to estimate the degree to which a compound will adsorb to soil and thus not migrate with groundwater. The higher the $K_{oc}$ value, the greater the tendency of the compound to partition into soil (OGE 1988). The sorption coefficient ($K_d$) is calculated from the $K_{oc}$ coefficient by multiplying the $K_{oc}$ value by the fraction of organic carbon in the soil.

Vapor pressure is a measure of the pressure at which a compound and its vapor are in equilibrium. The value can be used to determine the extent to which a compound would travel in air, as well as the rate of volatilization from soils and solution (OGE 1988). In general, compounds with vapor pressures lower than $10^{-7}$ mm mercury will not be present in the atmosphere or soil air in significant amounts, while compounds with vapor pressures higher than $10^{-2}$ mm mercury will exist primarily in the air (Dragun 1988). As **Table 5-1** shows, VOCs will exist primarily in the atmosphere and soil air, while pesticides and other SVOCs will exist in both the air and the soil.

The Henry's Law constant value ($K_H$) for a compound is a measure of the ratio of the compound's vapor pressure to its aqueous solubility. The $K_H$ value can be used to make general predictions about the compound's tendency to volatilize from water. Substances with $K_H$ values less than $10^{-7}$ atm-m$^3$/mol will generally volatilize slowly while compounds with $K_H$ greater than $10^{-3}$ atm-m$^3$/mol will volatilize rapidly (Lyman, Reehl, and Rosenblatt 1982).

Organic chemicals with differing chemical structures will biodegrade at different rates. Primary biodegradation consists of any biologically induced structural change in an organic chemical while complete biodegradation is the biologically mediated degradation of an organic compound into carbon dioxide, water, oxygen, and other metabolic inorganic products (Dragun 1988). The biodegradation rate of an organic chemical is proportional to the concentration:

$$-DC/Dt = kC^n ,$$

where:

C = concentration,
k = biodegradation rate constant = (2.303/t) log (a/[a-x]),
t = time,
a = initial concentration,
n = reaction order,
x = concentration at time t.

The half-life ($t_{1/2}$ = 0.693/k) is the time necessary for half of the chemical concentration to react. The biodegradation rate of an organic chemical is generally dependent on the presence and population size of soil microorganisms, capable of degrading the chemical.

Inorganic and organic constituents in soil, sediment, and groundwater are in continuous chemical and physical interaction with ambient surface and subsurface environmental conditions. The observed distributions of chemical concentrations in the environment are affected by these interactions and determine the chemical fate of these materials in the various media. Chemicals released into the environment are susceptible to several degradation pathways including hydrolysis, oxidation, reduction, isomerization, photolysis, photooxidation, biotransformation, and biodegradation. Transformation products resulting from these processes will behave distinctively in the environment.

### 5.3.1 Metals

Inorganic constituents detected in soil and sediment samples at WBG are associated with both the aqueous phase (unbound pore water) and with leachable metal ions on soil particles. The transport of these materials from unsaturated soils to the underlying groundwater is controlled by the physical processes of precipitation infiltration, chemical interaction with the soil, and downward transport of removed metal ions by continued infiltration. The additional physical mechanism of erosive transport is important for surface soil and sediment dispersal. The chemistry of inorganic interaction with percolating precipitation and varying soil conditions is complex and includes numerous chemical transformations that may result in altered oxidation states, ion exchange, adsorption, precipitation, or complexation. The chemical reactions, which are affected by environmental conditions including pH, oxidation/reduction conditions, and the type and amount of organic matter, clay, and the presence of hydrous oxides, may act to enhance or reduce the mobility and toxicity of the metal ions. In general, these reactions are reversible and add to the variability commonly observed in inorganics distributions in soil and sediment.

Inorganic chemical form impacts the solubility and mobility in the environment; however, chemical speciation is complex and difficult to delineate in routine laboratory analysis. Metals in soil and sediment are commonly found in several forms (Shuman 1991), including dissolved concentrations in soil pore water; metal ions occupying exchange sites on inorganic soil constituents; specifically adsorbed metal ions on inorganic soil constituents; metal ions associated with insoluble organic matter; precipitated inorganic compounds as pure or mixed solids; and metal ions present in the structure of primary or secondary minerals.

In situations where metal ions have been introduced to the environment by human activities, they are attributed to non-mineralogic occurrences. The dissolved (aqueous) fraction and its equilibrium fraction are of primary importance when considering the migration potential of metals associated with soil and sediment. Of the inorganic compounds that are likely to form, chlorides, nitrates, and nitrites are commonly the most soluble. Sulfate, carbonate, and hydroxides generally have low to moderate solubility. Soluble compounds are transported in aqueous form subject to attenuation, whereas less soluble compounds remain as a precipitate and limit the overall dissolution of the metal ions. The solubility of the metal ions also is regulated by ambient chemical conditions, including pH and oxidation/reduction.

The attenuation of metal ions in the environment can be estimated numerically using the retardation factor ($R_d$). The extent to which the velocity of the contaminant is slowed and is largely derived from the soil/water partitioning coefficient ($K_d$) is expressed by the following relation:

$$R_d = 1 + (K_d \rho_b)/\phi_w ,$$

**where:**

$\rho_b$ = the soil bulk dry density, (g/cm$^3$),
$\phi_w$ = soil moisture content, (dimensionless).

**Table 5-1** provides a range of $K_d$ values (as well as calculated $R_d$ values) for metal ions concentrations in various soil types. Because of variability in the media and in the methods of measurement, the reported $K_d$ values for metal ions vary over orders of magnitude and may similarly vary between samples. The $K_d$ values (as well as the $R_d$ values) presented in this table represent loam–type soil and are conservative because the site soil is more clay and silt than loam. Metal ion concentrations in the environment do not attenuate by natural or biological degradation because of low volatility and solubility of the ions. Metals concentrations may be biotransformed or bioconcentrated through microbial activity.

### 5.3.2 Organic Compounds

The predominant organic chemical compounds detected in WBG soils include explosives-related compounds and SVOCs (primarily PAHs). These compounds may be transformed or degraded in the environment by various processes, including hydrolysis, oxidation/reduction, photolysis, volatilization, biodegradation, or biotransformation. The half-life (time required to naturally reduce chemical concentration by one-half) of organic compounds in various media can vary from minutes to years depending on environmental conditions and the chemical structures of the compounds. Organic degradation may either enhance (through the production of more toxic byproducts) or reduce (through concentration reduction) the toxicity of a chemical in the environment. Biodegradation rates (Howard et al. 1991) for the detected organic compounds at WBG are shown in **Table 5-2**.

### 5.3.3 Explosives-Related Compounds

Explosives-related (nitroaromatic) compounds detected at WBG include 2,4,6-TNT, 2,4-DNT, 2,6-DNT, 1,3-DNT, and *o*-nitrotoluene, HMX, and RDX. The source, and nature and extent of the ammunition related constituents are discussed in Section 4.0.

Surface runoff process is predominantly an advective transport process that allows spatial and temporal variations of contaminants through washoff. Microbiological and photochemical transformations are two additional principal processes affecting the fate and distribution of munitions-related constituents in the environment. The additional chemical processes of oxidation/reduction and hydrolysis have not been observed to be important transformation mechanisms under environmental conditions with the possible exception of the slow hydrolysis of tetryl. Based on the results of culture studies involving the removal of TNT by activated sludge microorganisms, TNT has been concluded to undergo biotransformation but not biodegradation (Burrows et al. 1989). Photochemical transformations resulting from the effects of sunlight on explosive-related compounds are regarded as more important than microbiological transformations because of faster transformation rates.

TNT is readily biotransformed in the environment to amino, diamino, and azoxy compounds (Walsh 1990). It has been found (Funk et al. 1993) that the anaerobic metabolism occurs in two stages. The first stage is the reductive stage in which TNT is reduced to its amino derivatives. In the second stage, degradation to nonaromatic products begins after the reduction of the third nitro group. The biotransformation pathway for TNT in simulated composting systems proposed by Kaplan and Kaplan (1982) is shown in **Figure 5-2**.

The biotransformation of 2,4-DNT has been systematically studied in laboratory cell cultures. The pathway proposal for this biotransformation is shown in **Figure 5-3**. The reduction products includes the amino and azoxy derivatives as observed with TNT biotransformation. As with TNT and DNT, the principal mode of microbial transformation of the nitroaromatic compounds TNB and DNB is reduction of nitro groups to form amino groups.

## 5.4 FATE AND TRANSPORT ANALYSIS

Contaminant fate and transport modeling are based on the CSM for WBG discussed in Section 5.2. Based on the discussions presented in the CSM (Section 5.2), air and groundwater were not considered as important pathways for contaminant transport from WBG; therefore, neither air modeling nor groundwater modeling was performed for the WBG site. Contaminant levels in air are minor to nonexistent for both human and ecological receptors. Gaseous emissions are likely to be low because chemicals have had from 40 to 50 years to volatilize. Further, the ONG has no planned use for WBG until remediation is completed. Therefore, exposure to airborne chemicals is negligible. Surface water and sediment are the primary media for contaminant migration at this site. Modeling for these media is presented below. While groundwater is not considered an important contaminant transport pathway at this time, additional groundwater investigations are planned for WBG in the future.

### 5.4.1 Model Domain with Drainage Areas and Flow Components

The surface water flow and contaminant transport due to rainfall were modeled for WBG. The model domain with respect to the surrounding areas is shown in **Figure 5-4**. The CSM for surface water system was developed and understood to select the appropriate numerical model for a realistic representation of the physical system.

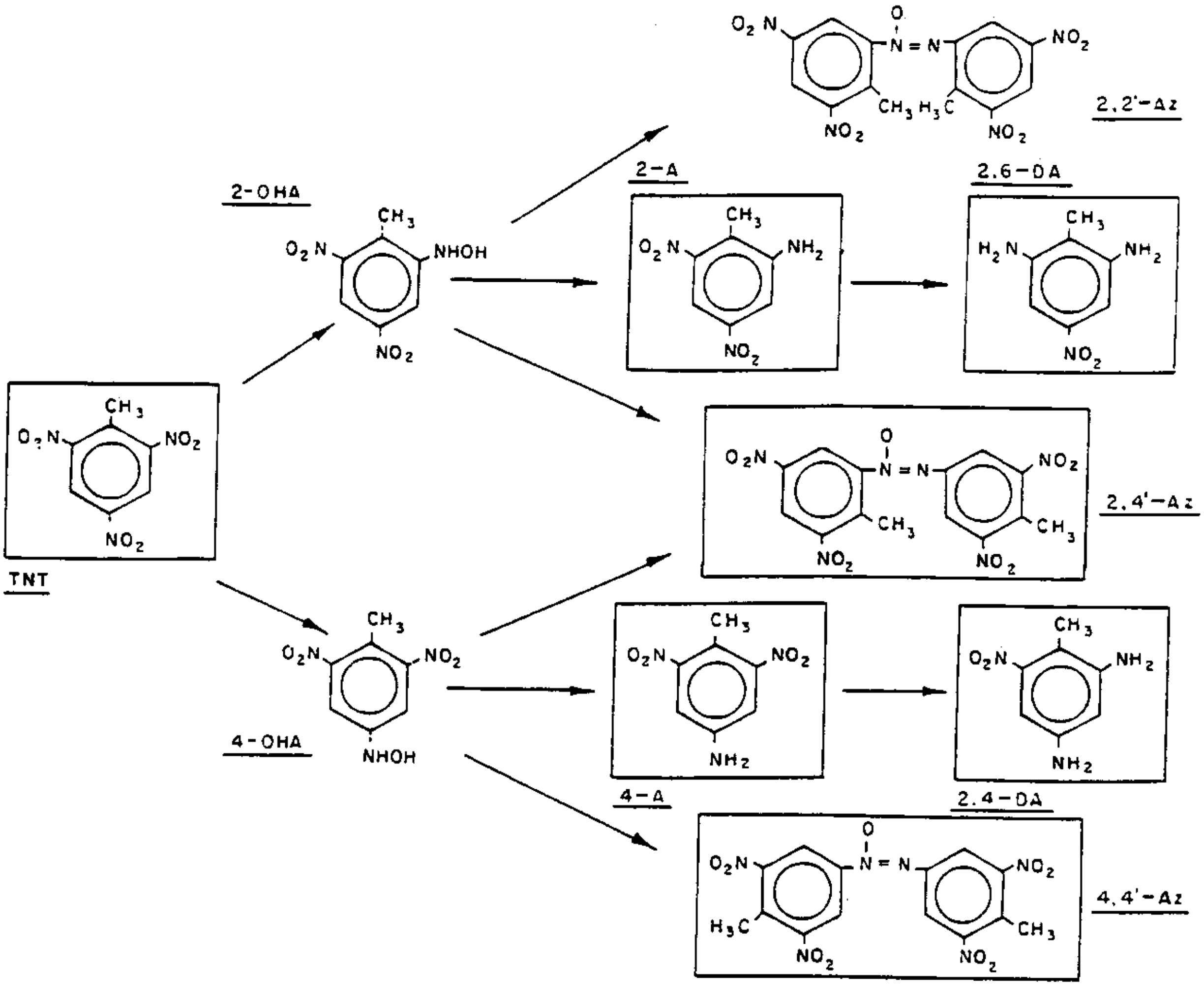


**Figure 5-2. Biotransformation Pathway for 2,4,6-Trinitrotoluene**
*Source:* Kaplan and Kaplan 1982

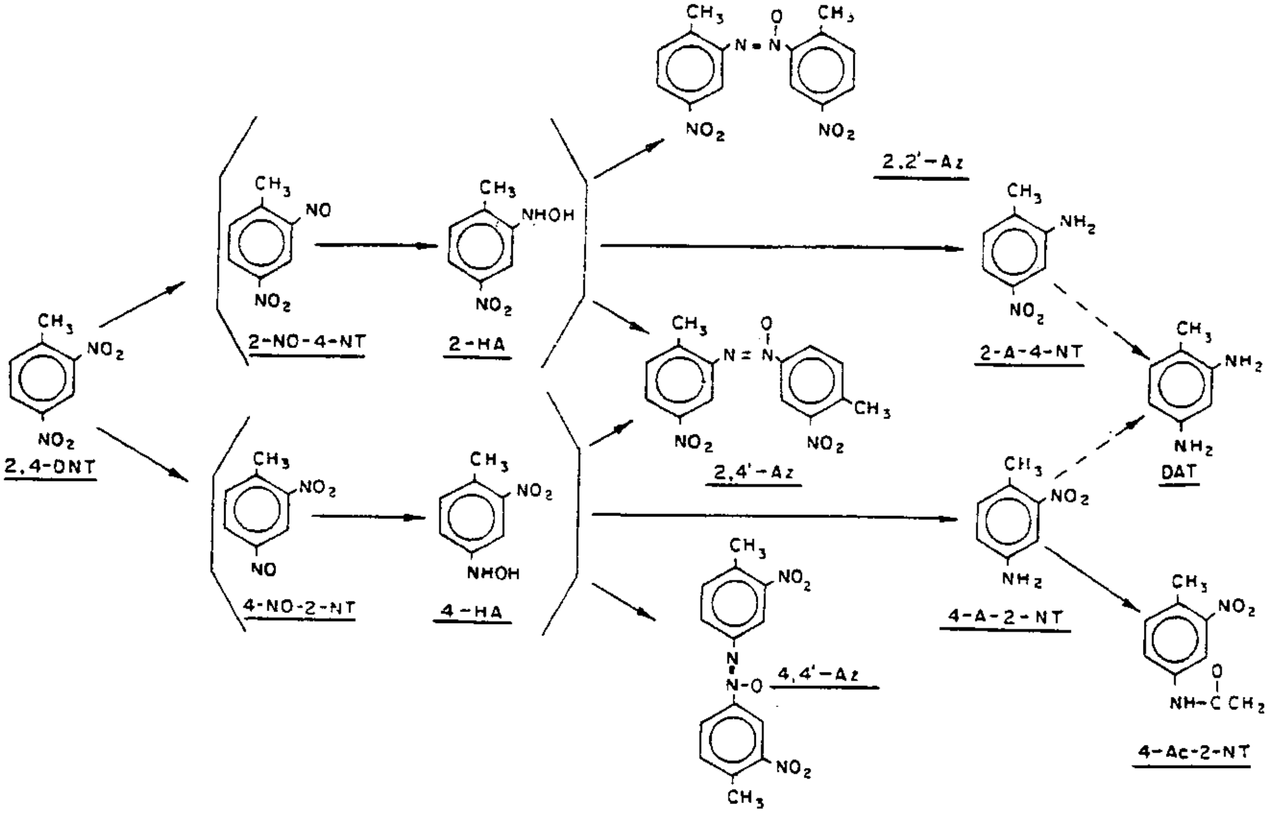


**Figure 5-3. Biotransformation Pathway for 2,4-Dinitrotoluene**
*Source:* McCormick et al. 1978

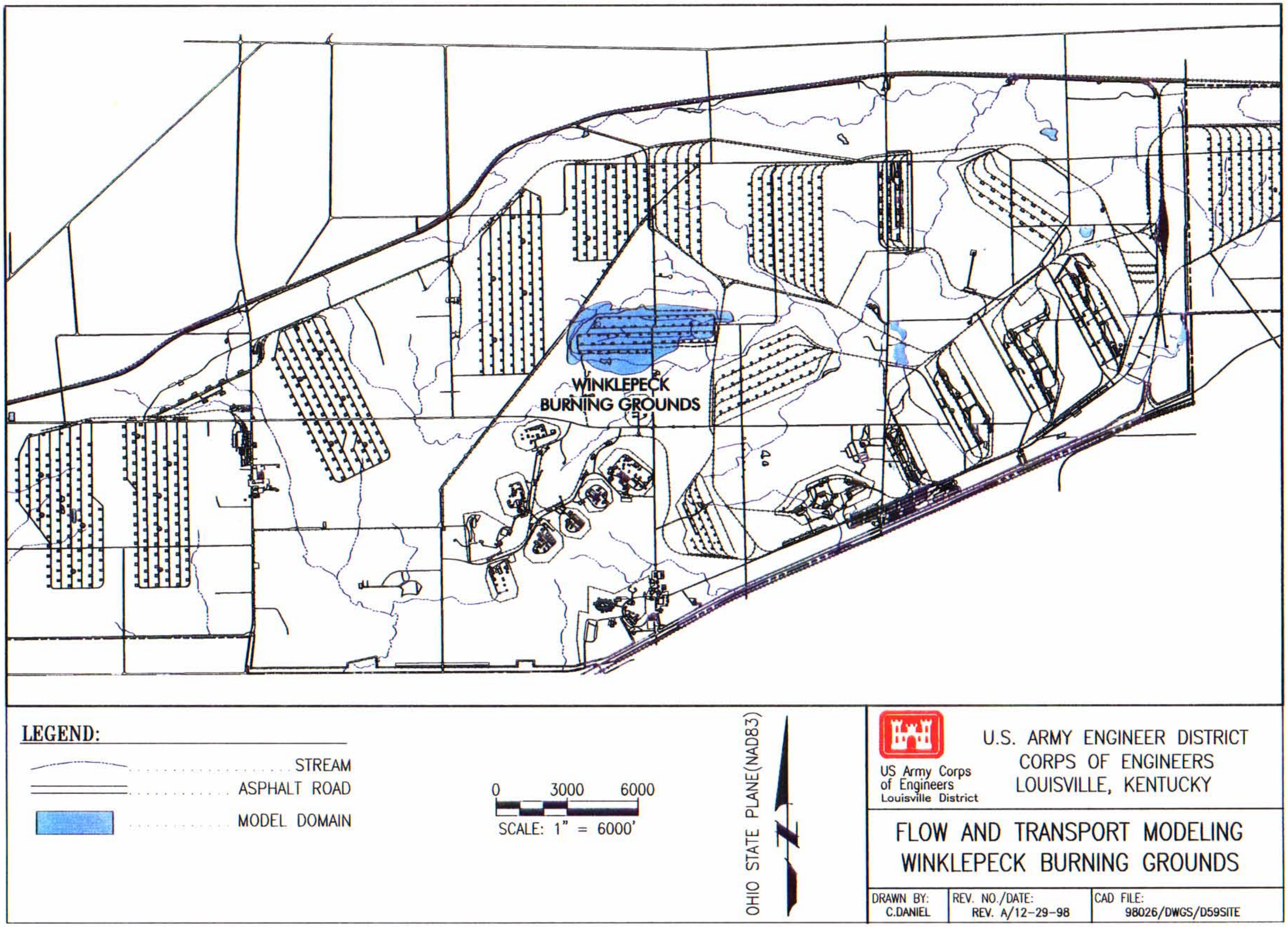


Figure 5-4. Location of Winklepeck Burning Grounds Modeling Site

Drainage areas are delineated to describe the movement and direction of overland flow to a drainage channel. These are also known as runoff areas. The topographic contours and the location of drainage ditches and tributaries help delineate drainage areas. The topographic divides are considered as the boundaries of the drainage areas contributing to drainage channels. The drainage areas were identified with relatively uniform hydrologic characteristics such as slope, roughness, imperviousness, etc. The drainage areas associated with tributaries/drainage ditches were identified to represent the flow and contaminant transport mechanism in the model domain.

The conceptualization of the physical system for washing off contaminants was based on identifying various hydrologic flow components including rainfall, infiltration, evaporation, and runoff. All these flow components with respect to the model domain gives a conceptual basis for hydrologic modeling of the physical system. The conceptual model thus generated for the model domain is shown in **Figure 5-5**.

A major flow component is the runoff flow that occurs due to rainfall excess. The rainfall excess is the amount of rainfall that remains after infiltration into the surface soil. The infiltration characteristics of the surface soil such as maximum infiltration capacity, minimum infiltration capacity, and infiltration decay constant were estimated to realistically conceptualize the hydrologic processes in the drainage areas.

The overland flow due to rainfall excess washes off contaminants and enters into drainage ditches and tributaries. The drainage ditches and tributaries route the flow to the drainage outlets. Spatially varying runoff flows are added from different drainage areas to different segments of drainage ditches and tributaries and finally routes along the course toward the outlet. The outlets in this model are shown in **Figure 5-5**.

### 5.4.2 Hydrologic Characteristics

Development of a conceptual model for overland flows and contaminant transport requires integration of a wide variety of hydrologic data. The data include drainage area characteristics, channel characteristics, climatic characteristics, contaminant source characteristics, etc. The hydrologic data required for conceptualizing the physical system in terms of runoff flow and contaminant transport are presented in brief in **Table 5-4**.

The drainage area characteristics were obtained from surface water divides, topographic contours, and surface soil conditions. Surface imperviousness data were estimated based on surface features in the drainage areas. The modeled domain is mostly pervious as bare ground surface even with low hydraulic conductivity clay soils is considered previous in the SWMM model. Imperviousness in the area was estimated based on the area covered by pads in each subcatchment. Drainage area slope, width of flow, depth of storage, etc. were estimated by evaluating topographic data and using empirical relationships.

The rainfall data were evaluated for the period 1949 to 1994. The rainfall data were obtained from the National Oceanic and Atmospheric Administration database. An average annual mean precipitation of 36.8 inches was obtained from the data. Rainfall in 1973 were closest to the average annual mean. Rainfall data from 1973 were used to simulate flow and transport for a typical condition as shown in **Figure 5-6**.

### 5.4.3 Model Selection

The model requirements included the ability to simulate time-varying hydrologic conditions, rainfall excess for runoff, infiltration, and contaminant transport in the system. The EPA Storm Water Management Model (SWMM; Huber and Dickinson 1988) was selected because it meets these criteria and because it has proven to be an efficient and robust numerical model.

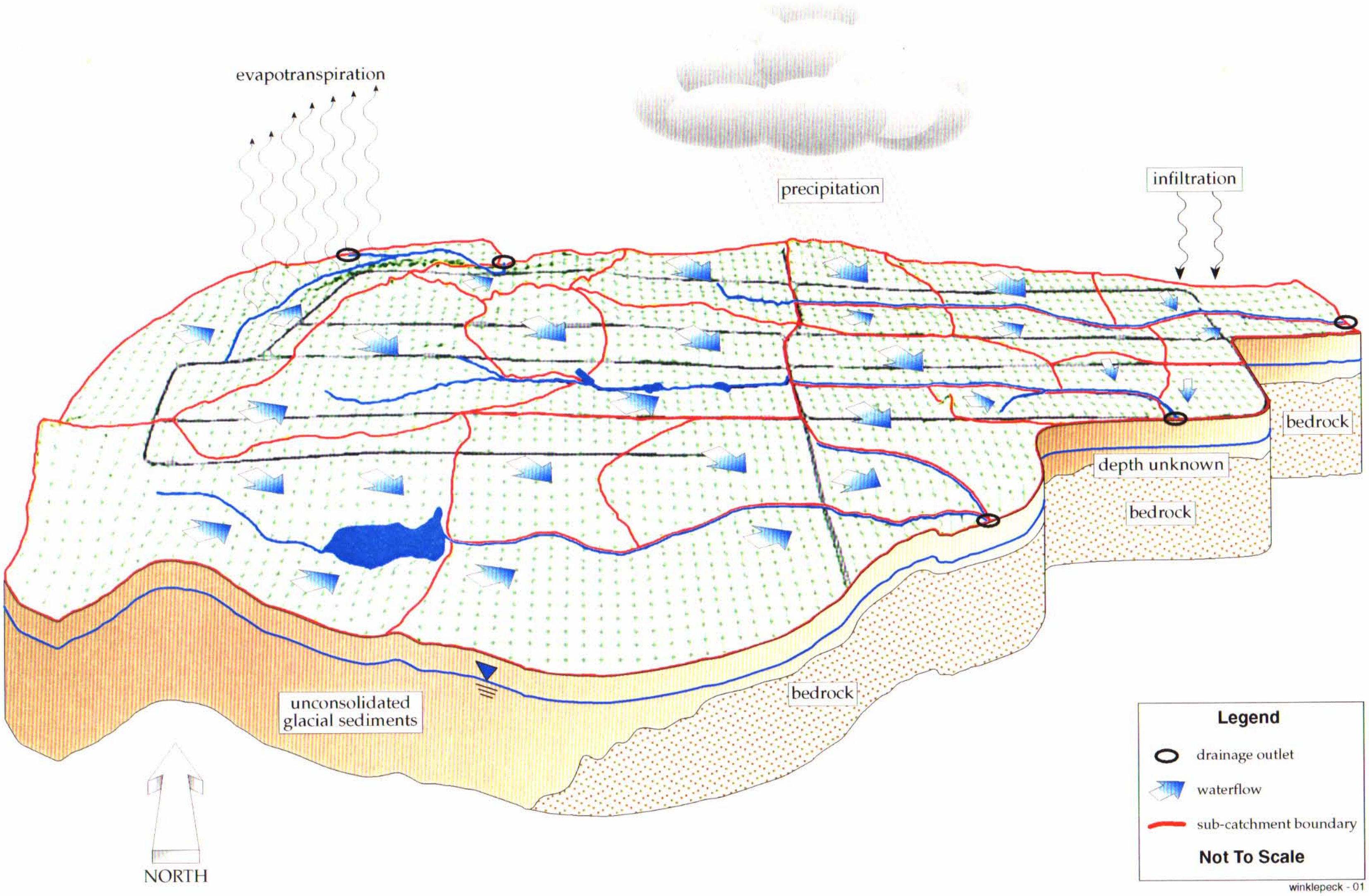
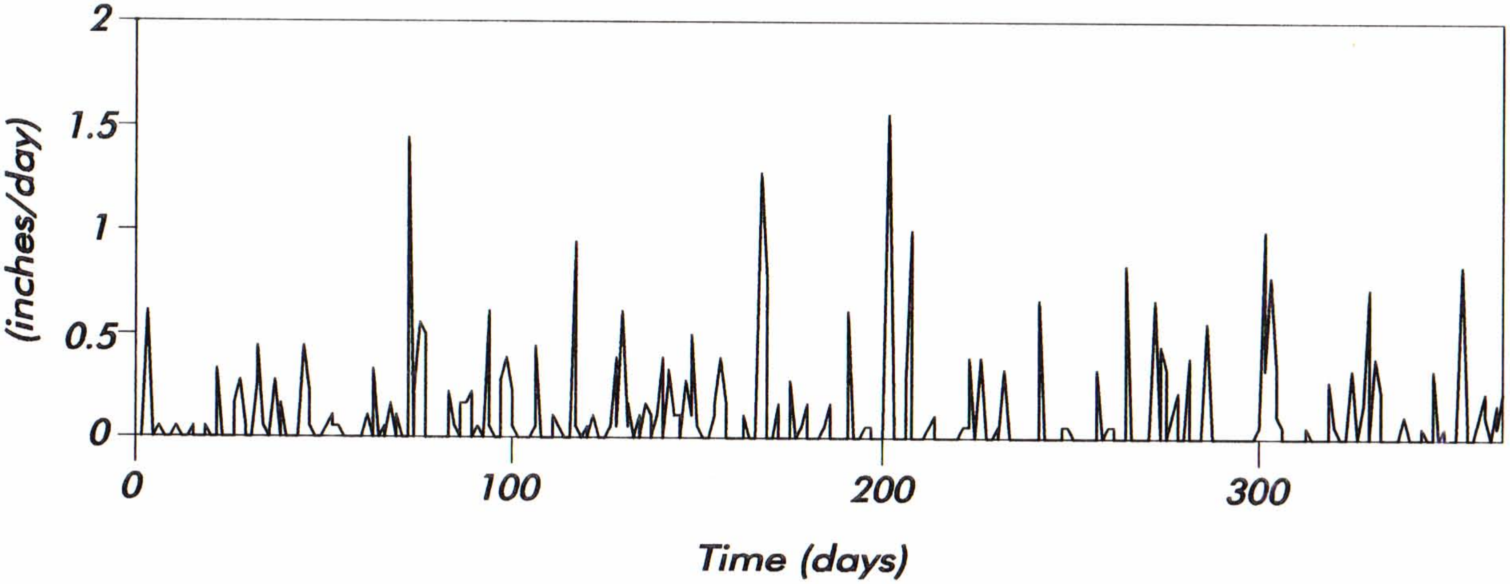


Figure 5-5. Conceptual Model Showing Drainage Areas for the Winklepeck Burning Grounds

### Table 5-4. Hydrologic Data Required for SWMM Model for the WBG Area

| Parameter | Value(s) | Source/Justification |
|---|---|---|
| *Drainage area characteristics* | | |
| Model domain (acres) | 319 | Based on the distribution of surface soil concentration |
| Drainage area (acres) | 4 to 36 | Based on topographic contour map showing surface water divides |
| Width of flow (ft) | 709 to 4,696 | Based on subcatchment in topographic contour map showing extent of overland flow |
| Slope % | 1 to 6 | Based on topographic contour map of the site |
| Maximum infiltration capacity (inches/hour) | 1 | Based on surface soil description |
| Minimum infiltration capacity (inches/hour) | 0.1 | Based on surface soil description |
| Surface imperviousness (%) | 0 to 14 | Based on evaluating surface characteristics on the site map. |
| Impervious area Manning's roughness coefficient | 0.013 | Obtained from SWMM model document for concrete/asphalt (Huber and Dickinson 1988) |
| Pervious area Manning's roughness coefficient | 0.45 | Obtained from SWMM model document for bluegrass sod (Huber and Dickinson 1988) |
| Impervious area depression storage (inches) | 0.02 to 0.07 | Based on formulation as a function of slope in SWMM manual |
| Pervious area depression storage (inches) | 0.1 | Based on evaluating surface characteristics on the site map |
| *Drainage ditch/tributary characteristics* | | |
| Manning's roughness coefficient | 0.04 | Obtained from SWMM model document for natural channel (Huber and Dickinson 1988) |
| Slope of drainage ditches/tributaries (%) | 0.1 to 9 | Based on topographic contour map |
| *Climatic characteristics* | | |
| Rainfall (inches) | 37 | Based on data of 1973, a typical rainfall year obtained from analyzing 1949 to 1994 data |
| Evaporation (inches/day) | 0.1 | SWMU default data used as an average value for the site (Huber and Dickinson 1988) |



US Army Corps
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LOUISVILLE, KENTUCKY

DISTRIBUTION OF RAINFALL
WINKLEPECK BURNING GROUNDS

| | | |
|---|---|---|
| DRAWN BY:<br>P. HOLM | REV. NO./DATE:<br>REV. A/12-30-98 | CAD FILE:<br>98036/DWGS/D59RAIN |

**Figure 5-6. Distribution of Rainfall for a Typical Rainfall Year**

The SWMM model was developed by Huber and Dickinson (1988) for the EPA. The SWMM is an overland and channel flow routing model that generates both hydrographs and pollutographs based on the hydrologic and contaminant transport characteristics. The model is organized in different blocks for simulating different hydrologic processes, which are described by Huber and Dickinson (1988).

The most often used blocks in SWMM are runoff, transport, and extended transport (extran). The overland runoff flow and contaminant transport in SWMM is computed by the runoff block. The transport block performs routing of flows and transport of contaminants in channels and conduits. The extran block performs routing of flows in channels and conduits including backwater effects. In the present modeling for WBG, the runoff block was used to route flows and contaminants toward the outlets.

The SWMM model has a wide range of use in simulating time-varying hydrologic conditions, rainfall excess for runoff, infiltration, washoff of contaminants from source areas to receptors, erosion, etc. for stormwater networks and waste discharge areas with drainage network. The model has been successfully used for a variety of projects (SAIC 1997; SAIC 1998; and University of Tennessee 1996).

### 5.4.4 Discretization

The flow and contaminant routing scheme in a hydrologic model is organized by discretizing the overall catchment into subcatchments or smaller drainage areas. The discretization is done on the basis of individual properties of subcatchments.

In SWMM, the subcatchments are evaluated as spatially lumped, nonlinear reservoirs with outflows routed through the channel or pipe network. The slope of the subcatchment is in the direction perpendicular to the width. Subcatchment-dependent data such as roughness coefficient, depression storage, etc., are also evaluated based on spatially varying conditions. The total drainage area of WBG is divided into 21 subcatchments and drainage ditches/tributaries, which are discretized into segments as shown in **Figure 5-7**.

### 5.4.5 Assumptions

Several assumptions in the conceptual model were made before the construction of the SWMM numerical model for the WBG area. The assumptions were necessary to bridge the gap between the available data and the requirements of the numerical model. Hydrologic characteristics such as surface roughness, slope, imperviousness, etc., for each subcatchment were taken as an average of the spatially varying values. It was assumed that the average values of the parameters were representative of the conditions in the subcatchment.

The hydraulic characteristics were assigned for different segments of the drainage ditches/tributaries. It was assumed that the segments are prismatic with constant slope and roughness. It was also assumed that the flow occurs in the direction of slope. The flow is assumed to occur from upstream to the downstream segments without any backwater effects.

The infiltration characteristics depend on the shallow groundwater table. The shallow aquifer and the unsaturated zone above it were assumed to have no effect on the infiltration and runoff processes. Capillary rise from the shallow groundwater table changes moisture conditions in the unsaturated zone. These changes have effects on the infiltration process, which in turn has an effect on runoff. In the SWMM model, Horton and Green-Ampt infiltration formulations are used with no solution for time-history of moisture content in the unsaturated zone. It is assumed that the effects of capillary rise above the groundwater table and time-varying moisture content in the unsaturated zone have negligible effects on the surface runoff-infiltration process.

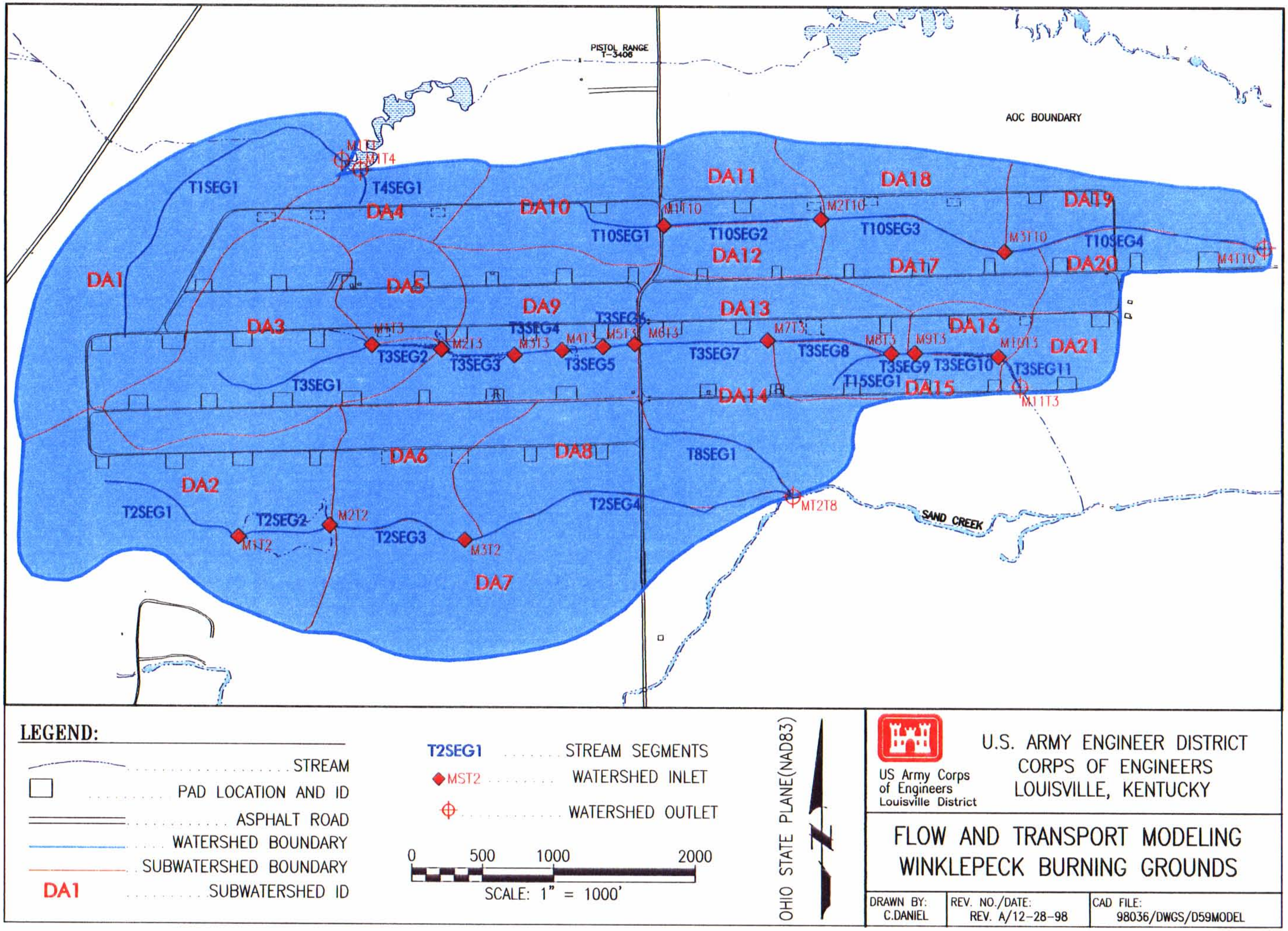


**Figure 5-7. Discretized Model Domain with Drainage Areas, Inlets, and Stream Segments**

There are assumptions involved in the SWMM model. The SWMM model assumes that the flow from subcatchments to the inlets does not occur through dynamic routing in both space and time. A lumped hydrologic routing technique is assumed for flow. The contaminant transport in SWMM is based on empirical buildup and washoff equations with initial loadings. The transport is not formulated using rigorous advection-dispersion partial differential equation based on continuity and advection-dispersion laws. The contaminant transport in surface water can be formulated based on advection-dispersion laws. The transport of contaminants by the bulk motion of the flowing water is known as advection. By advection law, the transport of contaminants in a flowing water is expressed as:

$$F_a = vC,$$

where:

$F_a$ = advective mass flow of contaminant per unit area ($M/(L^2T)$),
$v$ = flow velocity ($L/T$),
$C$ = concentration of contaminant ($M/L^3$).

The contaminants also spread out from the path of advective flow in a process known as hydrodynamic dispersion. According to dispersion law (known as Fick's first law), the transport of contaminant in water is:

$$F_d = D\nabla C,$$

where:

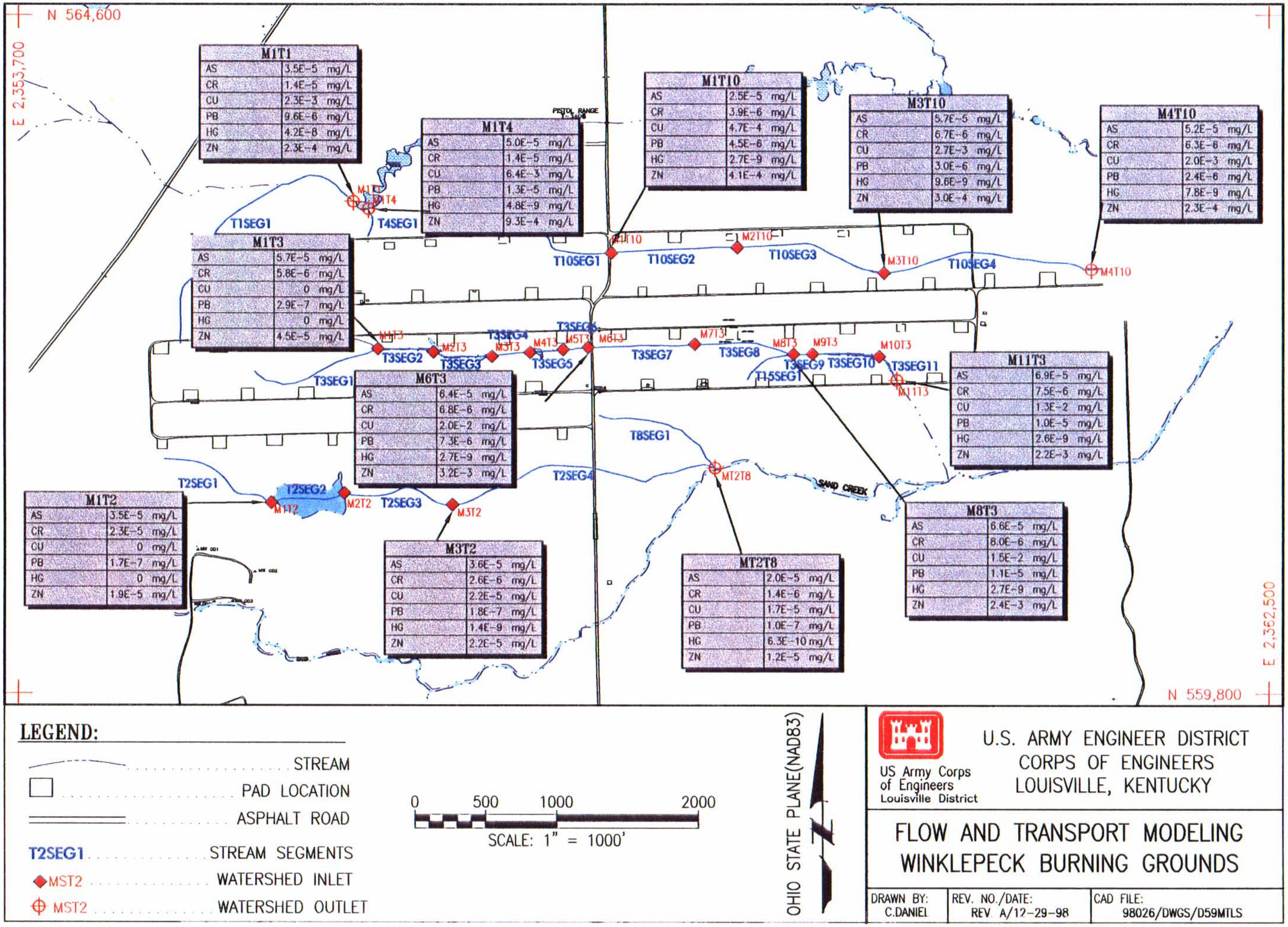
$F_d$ = dispersive mass flow of contaminant per unit area ($M/(L^2T)$),
$D$ = dispersion coefficient ($L^2/T$),
$\nabla$ = gradient operator.

Based on these advection-dispersion laws and equation of continuity, a parabolic partial differential equation can be developed to describe transport of contaminants in space and time in a flowing surface water system. In SWMM, the transport of contaminants is formulated using continuity criterion in conjunction with empirical expressions for buildup and washoff.

### 5.4.6 Model Results

The focus of the present modeling study was to determine spatial variations of contaminants for a typical rainfall year. Simulations were carried out for loading of contaminants from the drainage areas. As discussed in Section 4.54, SRCs were identified for the surface soil. However, to limit the number of model runs, only the representative contaminants from different chemical classes were selected. These chemicals are: arsenic, chromium, copper, lead, mercury, and zinc from the metal SRCs; benzo(a)pyrene, phenanthrene, and pyrene from the PAH SRCs; and TNB, TNT, DNT, HMX, and RDX from the explosive SRCs. The simulated concentrations of metals, organics, and explosives are shown in **Figures 5-8, 5-9**, and **5-10**, respectively. The simulated concentrations of contaminants largely depend on the available loadings from contaminated drainage areas.

The spread of metals for washoff due to runoff is shown throughout the model domain. This distribution of metals occurs due to the available loading throughout the site. However, the magnitudes of the concentrations of metals are not significant in any part of the model domain. The existence of more pervious areas in the model domain allows more infiltration with less runoff flow. Therefore, advective transport of contaminants due to runoff flow is not significant. An insignificant spread of organics and explosives is simulated as shown in **Figures 5-9** and **5-10**, respectively. These simulated distributions are consistent with the distribution of given loadings corresponding to the soil concentrations in the drainage areas with more pervious areas.



| M1T1 | |
|---|---|
| AS | 3.5E-5 mg/L |
| CR | 1.4E-5 mg/L |
| CU | 2.3E-3 mg/L |
| PB | 9.6E-6 mg/L |
| HG | 4.2E-8 mg/L |
| ZN | 2.3E-4 mg/L |

| M1T4 | |
|---|---|
| AS | 5.0E-5 mg/L |
| CR | 1.4E-5 mg/L |
| CU | 6.4E-3 mg/L |
| PB | 1.3E-5 mg/L |
| HG | 4.8E-9 mg/L |
| ZN | 9.3E-4 mg/L |

| M1T10 | |
|---|---|
| AS | 2.5E-5 mg/L |
| CR | 3.9E-6 mg/L |
| CU | 4.7E-4 mg/L |
| PB | 4.5E-6 mg/L |
| HG | 2.7E-9 mg/L |
| ZN | 4.1E-4 mg/L |

| M3T10 | |
|---|---|
| AS | 5.7E-5 mg/L |
| CR | 6.7E-6 mg/L |
| CU | 2.7E-3 mg/L |
| PB | 3.0E-6 mg/L |
| HG | 9.6E-9 mg/L |
| ZN | 3.0E-4 mg/L |

| M4T10 | |
|---|---|
| AS | 5.2E-5 mg/L |
| CR | 6.3E-6 mg/L |
| CU | 2.0E-3 mg/L |
| PB | 2.4E-6 mg/L |
| HG | 7.8E-9 mg/L |
| ZN | 2.3E-4 mg/L |

| M1T3 | |
|---|---|
| AS | 5.7E-5 mg/L |
| CR | 5.8E-6 mg/L |
| CU | 0 mg/L |
| PB | 2.9E-7 mg/L |
| HG | 0 mg/L |
| ZN | 4.5E-5 mg/L |

| M6T3 | |
|---|---|
| AS | 6.4E-5 mg/L |
| CR | 6.8E-6 mg/L |
| CU | 2.0E-2 mg/L |
| PB | 7.3E-6 mg/L |
| HG | 2.7E-9 mg/L |
| ZN | 3.2E-3 mg/L |

| M11T3 | |
|---|---|
| AS | 6.9E-5 mg/L |
| CR | 7.5E-6 mg/L |
| CU | 1.3E-2 mg/L |
| PB | 1.0E-5 mg/L |
| HG | 2.6E-9 mg/L |
| ZN | 2.2E-3 mg/L |

| M1T2 | |
|---|---|
| AS | 3.5E-5 mg/L |
| CR | 2.3E-5 mg/L |
| CU | 0 mg/L |
| PB | 1.7E-7 mg/L |
| HG | 0 mg/L |
| ZN | 1.9E-5 mg/L |

| M3T2 | |
|---|---|
| AS | 3.6E-5 mg/L |
| CR | 2.6E-6 mg/L |
| CU | 2.2E-5 mg/L |
| PB | 1.8E-7 mg/L |
| HG | 1.4E-9 mg/L |
| ZN | 2.2E-5 mg/L |

| MT2T8 | |
|---|---|
| AS | 2.0E-5 mg/L |
| CR | 1.4E-6 mg/L |
| CU | 1.7E-5 mg/L |
| PB | 1.0E-7 mg/L |
| HG | 6.3E-10 mg/L |
| ZN | 1.2E-5 mg/L |

| M8T3 | |
|---|---|
| AS | 6.6E-5 mg/L |
| CR | 8.0E-6 mg/L |
| CU | 1.5E-2 mg/L |
| PB | 1.1E-5 mg/L |
| HG | 2.7E-9 mg/L |
| ZN | 2.4E-3 mg/L |

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FLOW AND TRANSPORT MODELING
WINKLEPECK BURNING GROUNDS

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Figure 5-8. Distribution of Average Concentrations of Metals Based on Simulations for a Typical Rainfall Year

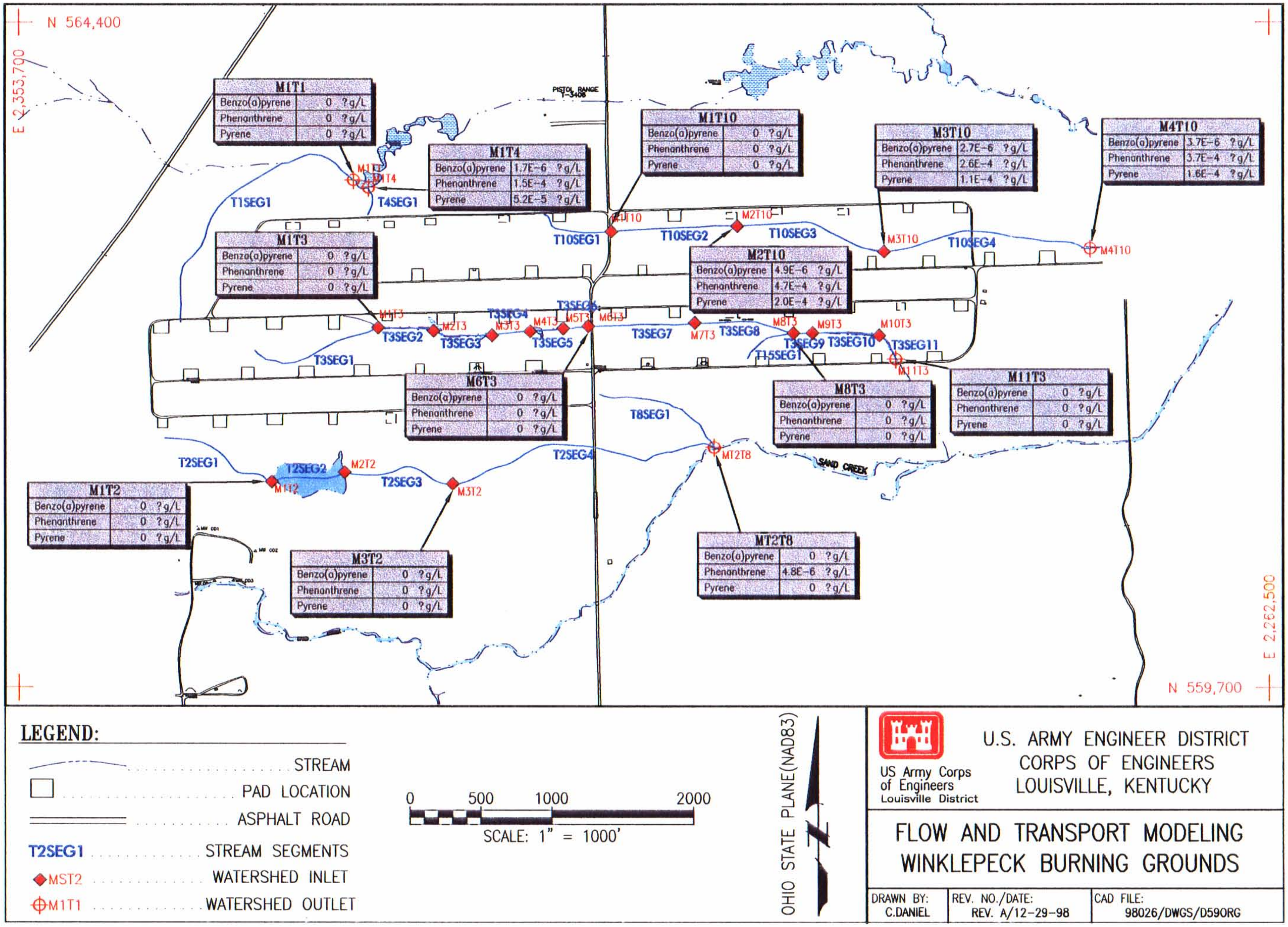
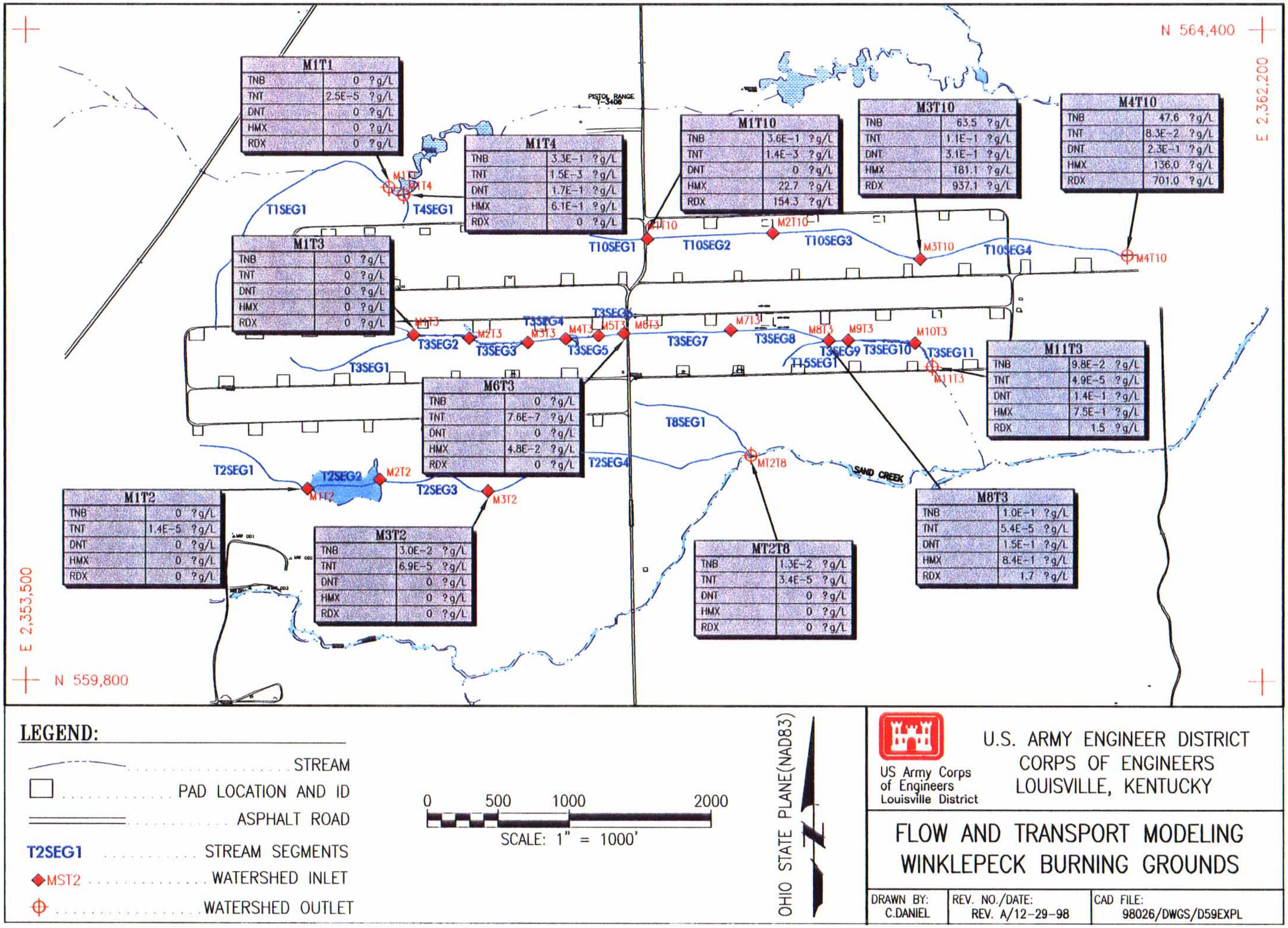


**Figure 5-9. Distribution of Average Concentrations of Organics Based on Simulations for a Typical Rainfall Year**



| M1T1 | |
|---|---|
| TNB | 0 ?g/L |
| TNT | 2.5E-5 ?g/L |
| DNT | 0 ?g/L |
| HMX | 0 ?g/L |
| RDX | 0 ?g/L |

| M1T4 | |
|---|---|
| TNB | 3.3E-1 ?g/L |
| TNT | 1.5E-3 ?g/L |
| DNT | 1.7E-1 ?g/L |
| HMX | 6.1E-1 ?g/L |
| RDX | 0 ?g/L |

| M1T10 | |
|---|---|
| TNB | 3.6E-1 ?g/L |
| TNT | 1.4E-3 ?g/L |
| DNT | 0 ?g/L |
| HMX | 22.7 ?g/L |
| RDX | 154.3 ?g/L |

| M3T10 | |
|---|---|
| TNB | 63.5 ?g/L |
| TNT | 1.1E-1 ?g/L |
| DNT | 3.1E-1 ?g/L |
| HMX | 181.1 ?g/L |
| RDX | 937.1 ?g/L |

| M4T10 | |
|---|---|
| TNB | 47.6 ?g/L |
| TNT | 8.3E-2 ?g/L |
| DNT | 2.3E-1 ?g/L |
| HMX | 136.0 ?g/L |
| RDX | 701.0 ?g/L |

| M1T3 | |
|---|---|
| TNB | 0 ?g/L |
| TNT | 0 ?g/L |
| DNT | 0 ?g/L |
| HMX | 0 ?g/L |
| RDX | 0 ?g/L |

| M11T3 | |
|---|---|
| TNB | 9.8E-2 ?g/L |
| TNT | 4.9E-5 ?g/L |
| DNT | 1.4E-1 ?g/L |
| HMX | 7.5E-1 ?g/L |
| RDX | 1.5 ?g/L |

| M6T3 | |
|---|---|
| TNB | 0 ?g/L |
| TNT | 7.6E-7 ?g/L |
| DNT | 0 ?g/L |
| HMX | 4.8E-2 ?g/L |
| RDX | 0 ?g/L |

| M1T2 | |
|---|---|
| TNB | 0 ?g/L |
| TNT | 1.4E-5 ?g/L |
| DNT | 0 ?g/L |
| HMX | 0 ?g/L |
| RDX | 0 ?g/L |

| M3T2 | |
|---|---|
| TNB | 3.0E-2 ?g/L |
| TNT | 6.9E-5 ?g/L |
| DNT | 0 ?g/L |
| HMX | 0 ?g/L |
| RDX | 0 ?g/L |

| M8T3 | |
|---|---|
| TNB | 1.0E-1 ?g/L |
| TNT | 5.4E-5 ?g/L |
| DNT | 1.5E-1 ?g/L |
| HMX | 8.4E-1 ?g/L |
| RDX | 1.7 ?g/L |

| MT2T8 | |
|---|---|
| TNB | 1.3E-2 ?g/L |
| TNT | 3.4E-5 ?g/L |
| DNT | 0 ?g/L |
| HMX | 0 ?g/L |
| RDX | 0 ?g/L |

Figure 5-10. Distribution of Average Concentrations of Explosives Based on Simulations for a Typical Rainfall Year

### 5.4.7 Discussion

In the present modeling study for the WBG area, the hydrologic and contaminant transport variables were integrated into SWMM for simulating a typical rainfall condition. Emphasis was placed on delineating drainage areas and discretizing drainage ditches/tributaries so that the site hydrologic and transport conditions were realistically represented for the site.

Physically meaningful data for surface roughness, depth of storage, washoff, and runoff parameters were used to develop a realistically accurate conceptual model for the WBG area. The topographic divides were identified from a 2-ft contour map to determine subcatchments that allow overland flow with contaminants to drainage ditches and tributaries. A typical rainfall year was selected based on the average rainfall for the site. The 14 primary SRCs in the surface soil and sediment that include metals, explosives, and PAHs were selected for simulation. Although they do not represent all the detected SRCs, they are representative of each class of chemicals at WBG with conservatism. The simulated results thus indicate representative conditions for the site.

The simulated spread of contaminants were not significant. The drainage areas representing overland flow planes for the WBG area are mostly pervious with grass cover. The overland flows and the associated velocities are not expected to cause significant advective transport of contaminants. The source areas were identified from the known conditions and observed data at the site. The loadings from subcatchments were based on observed data. Nonpoint source pollution at the site were not expected, and little sampling was done outside of the pads. For this, buildup of contaminants from nonpoint sources were not considered.

## 5.5 SUMMARY AND CONCLUSIONS

### 5.5.1 Fate and Transport Summary

Surface water is the only medium at the WBG considered for potential off-site contaminant migration. While groundwater is not considered an important contaminant transport pathway at this time, additional groundwater investigations are planned for WBG in the future. Therefore, modeling of contaminant fate and transport via surface water was performed. The findings of the surface water modeling are summarized below:

- A satisfactory and representative conceptual model for the WBG area for overland flow and contaminant transport was developed. Site-specific information for hydrologic and contaminant variables was used to develop the conceptual model for subsequent translation and integration into the SWMM numerical model. Six metals (e.g., arsenic, chromium, copper, lead, mercury, and zinc), five explosives (e.g., 1,3-TNB, 2,4,6-trinitrotoluene, 2,4-DNT, HMX, and RDX), and three PAHs [e.g., benzo(*a*)pyrene, phenanthrene, and pyrene], were modeled for the WBG site.

- The model simulation results were adjusted to match the reality based on transformation of observed peak soil concentration data of metals and organics through distribution coefficients at the site. The adjustments of parameters were done to achieve satisfactory agreement.

- The simulated distribution of contaminants for a typical rainfall year was not significant (i.e., predicted concentrations of contaminants were generally much below EPA or OEPA surface water criteria for human and ecological receptors) in the surface water system comprising overland flow from drainage areas to the drainage ditches and tributaries. Only, negligible amounts of explosives (TNB at 0.013 µg/L, and TNT at 0.000034 µg/L yearly average concentrations) from the WBG site were predicted to be discharging to the Sand Creek. Therefore, potential off-site contaminant migration via surface water and sediment pathways at WBG is not expected to be of any problem.

### 5.5.2 Fate and Transport Conclusion

- Based on SWMM modeling results, potential off-site (i.e., off WBG AOC) contaminant migration via surface water and sediment pathways at WBG is not expected to be any problem.

- Organic compounds and explosives detected at WBG subsurface soils are expected to degrade considerably before leaching to the groundwater. Potential off-AOC migration of these constituents via groundwater pathway is may be limited due to natural attenuation processes in the groundwater. However, site-specific data have not been gathered to confirm this.

- Heavy metals detected at WBG subsurface soil are not expected to significantly leach to the groundwater due to their high adsorption coefficients.