

# INTEGRATED MODELING OF DISULFOTON TRANSPORT AT AN ABANDONED LANDFILL SITE

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**ABSTRACT.** *Abandoned landfill sites pose an uncertain threat to water quality and environmental health. These sites, often located in rural areas, historically had little or no regulation for use or maintenance, and it is often unknown what hazardous materials may have been discarded. Evaluation and remediation of these sites can be extremely costly. Numerical models of contaminant fate and transport can be a cost-effective tool for assessing the potential hazards that a site may pose and for evaluating potential remediation measures, but seldom are models available which can simulate all aspects of contaminant fate and transport under complex site conditions. A study is presented describing the site investigation and remediation of an abandoned landfill reported to contain unknown quantities of disulfoton, an acutely toxic pesticide. Depending upon disulfoton concentration, it may or may not be dissolved in the soil water; thus, it may be transported in the soil either in an aqueous (dissolved) phase solution with soil water or as a non-aqueous phase liquid (NAPL). Because of the complex nature of the site and contaminant, it was necessary to employ three different numerical models to analyze different aspects of the problem. The article focuses particular attention on the use of different models for contaminant transport modeling in support of a Remedial Investigation (RI). Model selection was based on the contaminant phase and soil conditions. Various scenarios were hypothesized due to the uncertainty of input parameters. Procedures are presented for site remediation based on site investigation, numerical modeling and risk assessment.*

**Keywords.** *Contaminant transport, NAPL, Non-aqueous phase liquid, SHAW model, SWANFLO model, VAM2D model.*

The Snake River Plain Aquifer located in southern Idaho supplies vast quantities of water each year for drinking, recreation, industry, and irrigated agriculture. Leaching of pesticides and fertilizers due to recharge from irrigation may serve as a non-point source of groundwater contamination. However, waste disposal sites are a known and perhaps more significant source of groundwater pollution. Chemicals and waste products have historically been discarded at small landfill sites with little or no regulation. Many of these landfill sites have since been closed. One such site is the Murtaugh landfill. As with many abandoned landfill sites, the threat this site poses to water quality is uncertain.

The situation at the Murtaugh landfill is complicated. Moderate quantities of containers holding highly concentrated liquid disulfoton buried in trenches and heterogeneous layering of the underlying geological formation present several potential transport paths, i.e., movement downward through the vadose zone to ground water and laterally offsite above the low permeable layers. Also, the low water-solubility of disulfoton may

cause a dense non-aqueous phase liquid (NAPL) to travel along with the water-solute phase depending upon the leaking rate of disulfoton and the soil water content. A thorough understanding of the hydrogeologic, climatic and trench conditions coupled with modeling results for various scenarios aid in evaluating the fate and transport of the contaminant.

Numerical models can play an important role in evaluation and remediation of hazardous waste sites by quantifying the migration and fate of contaminants. However, it should be emphasized that modeling results must be interpreted with caution since they are dependent on assumptions and input conditions which may be uncertain. This article describes a case study of the site investigation and remediation of the Murtaugh landfill. The focus of this study is on the contaminant transport modeling conducted in support of a Remedial Investigation (RI) to assist in defining the baseline risk assessment and potential long-term extent of contamination as required by the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). Because no single model could simulate all aspects of the problem, a combination of three models was used to simulate contaminant fate and transport for different scenarios.

## BACKGROUND OF MURTAUGH LANDFILL

Murtaugh landfill is an approximately 6-acre (2.4-ha) tract within a 40-acre (16-ha) parcel of land owned by the U.S. Department of Interior Bureau of Land Management (USDI/BLM). The site is located about 3 miles (5 km) southwest of the town of Murtaugh in southern Idaho (fig. 1). Surface elevation of the landfill site is about 4330 ft (1320 m) above sea level and the surface slope is to

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**Figure 1—Geographical location of the Murtaugh landfill.**

the southeast at 3%. The nearest surface water features are Murtaugh Lake (2.5 miles or 4 km to the southeast) and the Snake River (3.5 miles or 6 km to the northeast). The regional aquifer is located in an interflow zone about 490 ft (160 m) deep within a basalt formation. A number of groundwater wells exist in the vicinity of the landfill site.

The site was operated as an open landfill with little or no regulation from the 1920s until 1972. Large quantities of pesticides reportedly disposed in several trenches at the site prompted the BLM to initiate a site inspection, which discovered low concentrations of pesticides and volatile organic compounds in soil samples (Weston, 1988). One drum sampled contained disulfoton, which established the presence of a RCRA (Resource Conservation and Recovery Act) acutely hazardous waste. Area water wells were sampled with no detection of contamination. Two test holes drilled to basalt indicated a caliche layer of varying thickness beneath the landfill site. An Expanded Site Investigation (ESI) was conducted by IT, Inc. (1990), which included hydrogeologic characterization, geophysical investigation and trench exploration at the site.

#### **HYDROGEOLOGY**

Soil surveys conducted above the bedrock formation within the 40-acre (16-ha) landfill site indicated that the top soil is about 1 ft (0.3 m) thick (Wells, 1989). Below the

top soil is a deep loess soil which rests upon the basalt formation and contains a relatively impermeable caliche layer, which is a hardened layer formed by the precipitation of calcium carbonate and other soil minerals. Soil analyses revealed that the loess layer is relatively homogeneous and contains approximately 25, 60, and 15% of sand, silt, and clay, respectively. Seismic refraction and resistivity surveys were conducted to configure the depths of caliche and basalt top. Seven boreholes were also drilled to the top of the basalt formation with soil samples collected at different depths. Results indicated the caliche layer is between 15 to 20 ft (4.6 to 6 m) below the soil surface and varies from a 2-ft (0.6-m) thick hard layer to scattered broken pieces or absent. The basalt top follows the surface topography closely at 20 to 30 ft (6 to 9 m) deep.

Geophysical loggings (IT, 1990) from the two groundwater wells (fig. 2) indicated that Snake River basalt is the dominant lithology between 30 ft (9 m) to over 500 ft (150 m) below the surface. An interconnected and highly fractured basalt exists between 30 and 120 ft (9 and 36 m) below the surface followed by very hard dense basalt. Three major interflow zones with clay, sand, and gravel deposits divide the basalt formation into distinctive basalt layers. Well logs from several irrigation wells in the vicinity of the landfill site also indicated the existence of the fractured basalt and three major interflow zones. A cross-sectional stratigraphic column beneath the landfill site is shown in figure 3. Geological conditions of the dense basalt were uncertain although columnar jointing structure could be seen in many places of the basalt plateau along the Snake River Canyon.

Two test wells drilled at the site extended to the depth of the regional aquifer. The uppermost aquifer is 490 ft (150 m) deep with over 40 ft (12 m) confined head at the two groundwater wells. No hazardous materials were detected from groundwater samples in the two test wells. Hydraulic conductivity of the interflow zones was determined to be 7.2 ft/dy (2.2 m/dy) (IT, 1990). Soil samples from boreholes were analyzed to obtain moisture content profiles and water retention curves.

The top soil at the site supports a wide variety of vegetation, such as cheat grass, goat beard, and sagebrush.

**Figure 2—Layout of the Murtaugh landfill.**

**Table 1. Disposal trench dimensions and estimated number of containers based on the Expanded Site Investigation (IT, 1990; WASTREN, 1992)**

Disposal Trench	Dimensions (m)			Estimated Containers		Container Type*
	Width	Length	Depth	Max No.	Min No.	
D-1	5.2	31.4	5.2	3975	1466	Drum
D-2	2.4	53.0	5.2	1320	437	Drum
D-3	5.2	15.5	5.2	1785	413	Drum
D-4	2.4	22.0	5.2	1186	300	Drum
D-5	2.4	22.0	5.2	1038	218	Drum
D-6	4.3	22.0	5.2	1779	507	Drum
D-7	3.0	29.6	5.2	2469	1970	Canister
D-8	1.5	22.0	5.2	6935	396	Canister
D-9	5.2	10.1	5.2	850	234	Drum
D-10	3.4	15.8	5.2	6172	3956	Canister
Total number of drums				14402	5545	
Total number of canisters				13107	4352	

\* Drum capacity is 55 gal (211 L); canister capacity is 5 gal (19 L).

the trench excavation, volumes of the liquid wastes in most of the containers remained uncertain.

**Figure 3—Geological materials beneath the Murtaugh site. (Depth not shown to proportional scale.)**

Canopy cover varies from season to season; canopy height was measured to be approximately 1 ft (30 cm). Residue cover on the surface was measured to be about 0.25 in. (0.5 cm) thick and estimated at 5,000 lb/acre (5600 kg/ha). Roots were observed to a depth of 3 ft (90 cm). Climate at the site is semi-arid with an average annual precipitation of about 10.3 in. (262 mm). Maximum monthly temperature occurs in July with an average value of 87°F (30°C) while minimum monthly temperature occurs in January at 16°F (-9°C). Snow and frozen soil are commonly present through the winters.

#### TRENCH EXCAVATION

Prior to trench excavation, geophysical surveys using electromagnetic, magnetic, and ground penetrating radar methods were conducted to determine the locations and geometries of the disposed wastes. Ten separate disposal trenches, designated as D-1 to D-10, were identified (IT, 1990) as shown in figure 2. To sample the buried materials, at least two exploration trenches were excavated parallel to or across the disposal trenches where large numbers of 55-gal (211-L) drums and 5-gal (19-L) canisters were tightly packed. Many of the metallic containers were corroded to rupture, and the liquid chemicals had leaked into the surrounding soils. A few buried containers were retrieved from each disposal trench for examination. Approximately 50% of the retrieved containers accommodated significant quantity of liquids and were estimated to contain 5 gal (19 L) per drum and 2.5 gal (9.6 L) per canister. Liquid samples from the containers and soil samples from the trench bottom were taken for analysis. Results showed that the liquid waste was mainly disulfoton at concentrations up to 790,000 ppm and that the soils were contaminated with concentrations up to 190 ppm. Trench dimensions and the estimated number of disposed containers are listed in table 1. Despite

#### DISULFOTON PROPERTIES

Disulfoton (chemical name: O,O-DiethylS-(2-ethylthioethyl) phosphorodithiolate) is an acutely toxic organophosphorus pesticide which was restricted for use in the United States in the early 1970s. It has a toxicity of 5.6 mg/kg (Ware, 1986) and its drinking water standard was set at 0.0003 ppm (U.S. EPA, 1992). Disulfoton will degrade primarily by oxidation reaction under the condition (pH = 7.8) within the Murtaugh site vadose zone. Other degradation mechanisms, such as hydrolysis, reduction, isomerization and microbial reactions, can also take place. Although disulfoton has a short half-life of 19 days (under ideal conditions), its oxidative metabolites are as toxic as disulfoton itself and somewhat more persistent in the soil environment. Table 2 lists some selected physicochemical and transport properties of disulfoton.

#### REMEDIAL INVESTIGATION

Results from the ESI demonstrated cause for concern. Thus, a Remedial Investigation (RI), was initiated to: determine the nature and extent of the contaminant; characterize the risk posed to the environment; and evaluate remediation measures to minimize risk. Portions of the RI included additional field investigations, including sampling and trenching, modeling of the contaminant fate and transport, baseline risk assessment, and recommendation of remediation alternatives (WASTREN, 1993). In the present article, we focus on modeling of the contaminant fate and transport.

**Table 2. Selected physicochemical and transport properties of disulfoton**

Physicochemical		Transport	
Molecular weight (g/mole)	274.4	log(K <sub>ow</sub> )	4.02
Specific gravity (g/cm <sup>3</sup> )	1.144	Viscosity (kg m <sup>-1</sup> s <sup>-1</sup> )	5.5 × 10 <sup>-4</sup>
Water solubility (ppm)	25	Diffusivity (m <sup>2</sup> /s)	5 × 10 <sup>-10</sup>
Vapor pressure (mm of Hg)	1.8 × 10 <sup>-4</sup>	Interfacial tension (dynes/cm)	
Volatility (mg/m <sup>3</sup> )	2.7	Air-water	72.75
Henry's constant (atm m <sup>3</sup> mole <sup>-1</sup> )	3.99 × 10 <sup>-6</sup>	Air-disulfoton	35.53
Half-life (days)	19	Disulfoton-water	37.22

## MODEL SELECTION AND PARAMETERIZATION

The site and contaminant properties at the Murtaugh landfill present a complex system to simulate. While potential routes of contaminant migration include vapors and airborne dust, modeling of the site focused exclusively on waterborne transport. Two conceivable transport paths exist for the contaminants to migrate outside the regulated landfill boundaries. Water percolating through the vegetative root-zone may drive the leaking chemical downward through the unsaturated zone to groundwater or laterally offsite above the low permeable caliche and dense basalt layers. Since disulfoton has a low water-solubility (25 ppm), a nonaqueous phase liquid (NAPL) could simultaneously exist with the water-solute phase, depending on the leaking rate of disulfoton and the soil water content. In addition to the NAPL flow, transport of disulfoton in aqueous phase must also be evaluated as its water-solubility is nearly five orders of magnitude higher than the drinking water standard. Because there are no reliable individual models that can simultaneously simulate a comprehensive surface water balance and interactive NAPL and dissolved chemical transport under the complex field conditions, a combination of three models was used to simulate different aspects of the problem.

Since a large percentage of the annual precipitation occurs in the winter months and frozen soil is commonly present during this period, the Simultaneous Heat and Water (SHAW) model, developed by Flerchinger and Saxton (1989a) and modified by Flerchinger and Pierson (1991), was used to predict evapotranspiration, percolation and runoff at the site. The two-dimensional, variably saturated models of SWANFLOW (Faust and Rumbaugh, 1989) and VAM2D (Huyakorn et al., 1989) were used to simulate the nonaqueous phase disulfoton and dissolved disulfoton migration separately. Transient percolation simulated by the SHAW model was applied to the ground surface as influx boundary conditions for the two contaminant transport models, providing the link between surface hydrologic processes and subsurface transport processes. A summary of the models used and the purpose for each is given in table 3.

The rate at which the buried canisters would fail was uncertain but trench excavation revealed that some had already corroded to rupture. Walton et al. (1989) studied the release of contaminants at the Idaho National Engineering Laboratory (INEL) under similar climatic and soil conditions. Based on these results, it was conservatively assumed that the buried containers would fail over a five-year period, during which all the chemicals currently present would leak out into the trenches. The estimated volume of disulfoton in each disposal trench (table 1) was used to calculate a constant leaking rate which was assigned to the corresponding trench-bottom as

a steady-state influx boundary condition. All simulations were proceeded for 120 years which was divided into a 20-year pre-leaking period to allow assumed soil moisture gradients to equilibrate with the existing climate, a 5-year leaking period, and a 95-year redistribution period. Observations of soil water content taken during the site investigation were sparse, but were used to ensure that the models approximately represented soil moisture at the site.

### SURFACE WATER BALANCE — SHAW MODEL

The SHAW model simulates the water balance of a one-dimensional profile from the top of plant canopy to a specified depth within the soil accounting for precipitation, snow accumulation and melt, evapotranspiration, percolation, runoff, and soil water content. It has detailed provisions for soil freezing and thawing and has been tested under a variety of site and climate conditions (Flerchinger and Saxton, 1989b; Flerchinger and Hanson, 1989; Flerchinger and Pierson, 1991; Pierson et al., 1991; Xu et al. 1991). The model simulates transport of solutes and includes provisions for first-order degradation of solutes, transport of degradation products, and partitioning of solutes between absorbed and solution phases.

Leaf area index for the plant canopy at the site was assumed to vary seasonally from 0.25 to 1.50. Weather conditions input to the model were supplied by CLIGEN, the climate simulation model (Lane and Nearing, 1989), which was used to generate an extended 100-year period of daily weather data for the Burley Weather Station, located 20 miles (30 km) east of Murtaugh. A unit gradient boundary condition was set on the bottom of the 30-ft (9-m) simulated profile. Since SHAW simulates dissolved chemical transport in one-dimension, the total volume of disulfoton was assumed to be spread uniformly over the 100 ft × 300 ft (30 m × 100 m) trench area at a depth between 11.5 to 18.0 ft (3.5 to 5.5 m). Deep percolation calculated by the SHAW model was used as input to the other models.

### NAPL TRANSPORT — SWANFLOW

Modeling contaminant transport in an air-NAPL-water three-phase system requires the functional relations of capillary pressure and relative permeability with saturation, which rarely exist for most of the organic chemicals and geologic materials in practice. SWANFLOW adopts a method (Stone, 1973; Aziz and Settari, 1979) to estimate the fluid behavior of a three-phase system using functional relations of air-water, air-NAPL, and NAPL-water two-phase systems. The extension from two-phase to three-phase systems is approached by regarding the total liquid saturation in such three-phase systems to be a function of the interfacial pressure at the air-liquid interface, independent of the number of liquids contained in the porous medium. Parker et al. (1987) proposed a technique

**Table 3. Summary of model usage at the Murtaugh Landfill Site**

Model	Processes Simulated	Use at Murtaugh Landfill	Region Simulated
SHAW	Surface energy and water fluxes; 1-D energy, water and solute transport	Surface water balance; boundary conditions for other models	Loess, caliche and fractured basalt layers
VAM2D	2-D water and solute transport	Transport of dissolved phase disulfoton	Loess, caliche, basalt, and interflow layers
SWANFLO	2-D water and non-aqueous phase liquid (NAPL) flow	Transport of NAPL disulfoton	Loess and caliche layers

to scale the capillary pressure-saturation relations for two-phase systems so that the fluid-dependent and porous medium-dependent factors can be explicitly distinguished and a single parametric expression can be used for different paired fluid systems. The scaling process is achieved through the following transformations:

$$S_w^{aw} (\beta_{aw} h_{aw}) = S_w^{ow} (\beta_{ow} h_{ow}) = S_w^{ao} (\beta_{ao} h_{ao}) \quad (1)$$

where the superscripts and subscripts of a, o, and w represent air, NAPL and water of the air-water, NAPL-water and air-NAPL two-phase systems, respectively.  $S$  is the effective saturation of the wetting fluid;  $h$  is the capillary pressure;  $\beta$  is the fluid pair-dependent scaling factor and can be calculated from ratios of fluid interfacial tensions (Lenhard and Parker, 1987). Values of  $\beta_{ow}$  and  $\beta_{ao}$  were calculated to be 1.95 and 2.05 with the two-phase interfacial tensions, respectively. Applying the scaled capillary pressure-saturation relations to the Van Genuchten function (Van Genuchten, 1980) gives a parametric expression as:

$$S_j^{ij} = \{1 + [\alpha (\beta_{ij} h_{ij})]^n\}^{-m} \quad (2)$$

Therefore, capillary pressure-saturation relations in the NAPL-water and air-NAPL systems of a porous medium could be determined if the soil water retention curve is available. This scaling method was tested and found to agree well with experimental data (Busby and Rolston, 1992).

Because data appropriate for the SWANFLOW model was not available for the basalt, the SWANFLOW model was applied to characterize the NAPL flow to a depth of 25 ft, which included the loess and caliche layers above the fractured basalt. Hydraulic properties for the silt loam and caliche used in the model are listed in table 4. Residual saturation of NAPL (disulfoton) in the soil was assumed to

**Table 4. Hydraulic properties for the silt loam, caliche and basalt used in simulations**

Property	Loam Soil*	Caliche†	Fractured Basalt*	Dense Basalt‡	Interflow Zones*
Conductivity ft/dy [m/dy]	2.6 [0.8]	0.0003 [0.0001]	7.0 [2.1]	0.00005 [0.00002]	1.0 [2.2]
Porosity (%)	50	25§	10	1	10
Air entry ft of H <sub>2</sub> O [m]	-1.0 [-0.3]	-22 [-6.8]	-1.0   [-0.3]	-1.0   [-0.3]	-1.0 [-0.3]
$\alpha$ ft <sup>-1</sup> [m <sup>-1</sup> ]	0.16 [0.52]	0.38 [1.26]	0.07   [0.23]	0.07   [0.23]	0.047 [0.23]
$n$	2.67	1.7	1.61	1.70	1.61
Organic carbon (%)	0.32	0.001	NA	NA	NA

\* Measured or recommended by WASTREN (1992); unsaturated parameters of loam soil based on water retention data summarized by DBS&A (1992).

† Estimated based on Faust and Rumbaugh (1989) and Evans and Nicholson (1987).

‡ Adopted from Freeze and Cherry (1979).

§ Based on measurement of caliche material.

|| Based on values used by Deng and Flerchinger (1994).

be equal to the residual saturation of water (Lenhard and Parker, 1987). Boundary conditions included a constant matric potential (water saturation) boundary condition applied at the bottom of the simulation domain where the water saturation was measured to be 0.3, and time-dependent percolation based on the SHAW simulation introduced as an upper boundary. Constant NAPL leakage at the bottom of the disposal trenches was also applied during the five-year leaking period.

#### AQUEOUS PHASE TRANSPORT — VAM2D

VAM2D is a variably saturated finite element groundwater flow and solute transport model in two dimensions (Huyakorn et al., 1984, 1985). The saturated/unsaturated flow equations are approximated using the Galerkin finite element method. The contaminant transport simulation can account for advection, mechanical dispersion and molecular diffusion. The model uses linear equilibrium process for adsorption and desorption, and first order reaction mechanism for chemical or biological degradations. VAM2D is somewhat limited in that it considers the molecular diffusion transport to be constant within each porous material. In fact, diffusion transport in the soil depends on tortuosity of the transport path, which is affected by soil water content. Thus, modification was made to VAM2D for this application to account for the dependence of diffusion transport on soil water content (Bresler, 1973; Campbell, 1985).

Unlike the SWANFLOW simulations, dissolved chemical migration was simulated from the ground surface to the top of the uppermost aquifer because hydraulic properties for dissolved chemical transport in the basalt formation were better defined. Murtaugh landfill is located in the Snake River Plateau where hydrogeologic properties of the basalt formation have been widely studied at the INEL site approximately 100 miles (170 km) northeast of the Murtaugh landfill (Bishop, 1991; Knutson et al., 1992; TerraTek Inc., 1988). Therefore, hydraulic properties of the fractured basalt and dense basalt from the INEL area were adopted for the VAM2D simulations as presented in table 4.

Contaminant dispersivity is scale dependent (Fried, 1975; Gillham et al., 1984; Pickens and Grisak, 1981) and has a trend for larger values to be associated with larger contamination zones. Fedler et al. (1989) showed that the tracer migration simulation using a longitudinal dispersivity of 2.5 ft (0.76 m) agreed well with field measurements with a contamination zone over 165 ft (50 m) long. A longitudinal value of 1.5 ft (0.46 m) was used in the VAM2D simulations because the contamination zone at the Murtaugh site was restricted by the low permeable caliche layer and dense basalt. Boundary conditions for the VAM2D simulations were similar to those used in the SWANFLOW runs, i.e., 100-year, time-dependent percolation from SHAW was introduced at the surface.

#### MODEL RESULTS

The surface water balance simulated by SHAW indicated that the annual evapotranspiration averaged 10.3 in. (262 mm) from the Murtaugh landfill, which accounted for most of the precipitation. Runoff averaged 0.4 in. (11 mm) annually and was most often associated with snowmelt and frozen soil. Percolation was zero most

of the time, except for a few wet years which accumulated approximately 0.25 in. (8 mm) of percolation over 100 years, which was used as an upper boundary condition for the SWANFLOW and VAM2D runs. Simulated soil water contents for the three models fell within the range of values for samples collected in the loess soil to a depth 26 ft (8 m).

Because of the uncertainty in site and contaminant conditions, several scenarios were examined to determine the effect of the caliche layer, contaminant volume and degradation. Four scenarios were examined: (1) maximum estimated volume of contaminants; (2) maximum estimated volume of contaminants with no caliche layer present; (3) minimum estimated volume of contaminants; and (4) maximum estimated volume of contaminants with degradation of contaminants considered.

#### **SCENARIO 1: MAXIMUM ESTIMATED VOLUME OF CONTAMINANTS**

This scenario assumed that the disposal trenches accommodated the maximum number of containers (14,402 drums and 13,107 cans). According to WASTREN (1992), half of the drums and canisters contained 2.5 and 1.25 gal (9.6 and 4.8 L) of disulfoton, respectively. Although concentrations up to 790,000 ppm were measured, the commercial grade of 940,000 ppm was assumed for simulations. A 2-ft (0.6-m) thick caliche layer was assumed to exist throughout the lateral extent. This scenario was to evaluate downward movement of disulfoton with the maximum estimated amount of buried chemicals.

Figure 4 shows the concentration profiles from the SHAW simulation at end of the five-year leaking period and after 10 years. Much of the simulated transport occurred during the five-year leaking period, and based on the symmetry in disulfoton concentration, diffusive transport was the dominant mode of transport. The maximum concentration was approximately 5,000 ppm, much higher than disulfoton water-solubility, indicating that soil water was insufficient to dissolve the leaking chemical and that a non-aqueous phase (NAPL) disulfoton would exist. Both SHAW and VAM2D simulations indicated that disulfoton would penetrate through the caliche layer and reach the top of the fractured basalt. However, the VAM2D simulation indicated that the dissolved disulfoton was not able to travel further down to the top of the dense basalt within the 100-yr simulation

**Figure 4—Concentration profile of disulfoton simulated by the SHAW model with maximum estimated amount of contaminant (scenario 1).**

because the relatively impermeable caliche prevented large volumes of dissolved chemical from penetrating through it. Concentration at the plume center was higher than the disulfoton water-solubility, indicating the existence of non-aqueous phase liquid in the soil environment, which would be more correctly addressed by the SWANFLOW model.

Figure 5 shows disulfoton saturation contours from the SWANFLOW simulation at the end of the five-year leaking period. Disulfoton saturation is defined as the fraction of pore space occupied by pure disulfoton. Results indicated that the caliche layer completely stopped the downward movement of NAPL causing disulfoton to be perched above the caliche layer. Thus, diffusion of dissolved chemical through the caliche layer, as simulated by the SHAW and VAM2D models, appears to be a more efficient means of transport across the caliche layer. Lateral flow was insignificant under the assumed conditions. One hundred years after leaking, the NAPL disulfoton had not moved appreciably.

#### **SCENARIO 2: NO CALICHE LAYER PRESENT**

This scenario was created to quantify disulfoton migration without the deterrent of caliche layer. Each disposal trench was assumed to accommodate the maximum numbers of containers as in scenario 1. Figures 6 and 7 show the concentration contours of the simulation at the end of the five-year leaking period and the end of the 100-year redistribution period from the VAM2D simulation. Without preclusion of the caliche layer, dissolved chemicals simulated by the VAM2D model reached the fractured basalt easily and then moved deeper to the top of the dense basalt at a depth of 120 ft (36 m) depth. Large concentrations of the

**Figure 5—Disulfoton saturation ( $m^3/m^3$ ) contours simulated by the SWANFLOW model at end of the five-year leaking period with maximum estimated amount of contaminant (scenario 1).**

**Figure 6—Concentration contours (ppm) of disulfoton simulated by the VAM2D model at end of the five-year leaking period with maximum estimated amount of contaminant and no caliche layer (scenario 2).**

**Figure 7–Concentration contours (ppm) of disulfoton simulated by the VAM2D model after 100 years with maximum estimated amount of contaminant and no caliche layer (scenario 2).**

chemicals remained near the bottom of the trenches even 100 years after leaking.

Figures 8 and 9 show the disulfoton saturation contours within the soil system. SWANFLOW simulations indicated that NAPL disulfoton could easily reach the fractured basalt beneath disposal trenches D-8, D-7 and D-10, which contained larger numbers of containers, by gravitational movement. However, NAPL flow became insignificant when disulfoton saturation decreased to or below the residual saturation. Residual disulfoton would then serve as a long term contaminant source and would eventually degrade or dissolve into the water phase.

#### **SCENARIO 3: MINIMUM ESTIMATED VOLUME OF CONTAMINANTS**

The volume of buried disulfoton is a significant factor in contaminant transport. This scenario was run with the minimum estimated amount of disulfoton indicated in

**Figure 8–Disulfoton saturation ( $m^3/m^3$ ) contours simulated by the SWANFLOW model at end of the five-year leaking period with maximum estimated amount of contaminant and no caliche layer (scenario 2).**

**Figure 9–Disulfoton saturation ( $m^3/m^3$ ) contours simulated by the SWANFLOW model after 100 years with maximum estimated amount of contaminant and no caliche layer (scenario 2).**

table 1 to assess the significance of chemical quantity on solute transport. According to WASTREN (1992), 50% of the drums and cans were assumed to contain 0.5 gal (1.9 L) of concentrated disulfoton. These assumptions resulted in a scenario with about 21 times less disulfoton than scenario 1. Other conditions of this simulation were the same as scenario 1. Under these conditions, concentration profiles from the SHAW and VAM2D models were significantly lower, the contaminated zone was smaller and downward movement of disulfoton was less compared to scenario 1. Concentrations at the center of the plume were not significantly above the solubility limit of 25 ppm, suggesting that NAPL flow is not significant in this scenario. With the deterrence of the caliche layer, the dissolved chemical was not able to reach the basalt formation. Even after 100 years, contaminant migration simulated by VAM2D did not extend significantly beyond the bottom of the trenches and concentrations were generally below 0.01 ppm.

#### **SCENARIO 4: DEGRADATION OF CONTAMINANTS CONSIDERED**

The assumption of no degradation in the previous scenarios is not realistic for disulfoton. The half-life of disulfoton under optimum reaction conditions is 19 days. However, within the soil at depths near the bottom of the trenches where oxygen and water may be limiting, the actual half-life would be somewhat longer. Studies have been performed to quantify the fate of disulfoton applied to near-surface soils, which reported a half-life of 50 days for disulfoton and its oxidative metabolites (Howard, 1991). A value of 190 days, 10 times longer than the optimum half-life, was assumed for soil conditions surrounding the trenches. For the SHAW simulation, one third of the disulfoton undergoing transformation was assumed to be subject to hydrolysis, and two-thirds followed the oxidation reaction producing toxic metabolites. Degradation of metabolites is not considered by the SHAW model. Figure 10 shows the concentration profile of disulfoton and its toxic oxidative metabolite at the end of the 10-year simulation period based on the SHAW simulation. Obviously, disulfoton concentration drops to a very low level while its toxic metabolites remain at high concentration. Similarly, maximum disulfoton concentration from the VAM2D simulation at the end of

**Figure 10–Concentration profile of disulfoton and its oxidative metabolite simulated by the SHAW model 10 years after the beginning of the five-year leaking period (scenario 4).**

the five-year leaking period was less than 25 ppm, and disulfoton disappeared completely a few years afterward.

## SUMMARY OF REMEDIAL INVESTIGATION

Field activities conducted as part of the RI included excavation of 11 exploratory trenches to sample product from the containers and soil within the trenches, and to better ascertain the volume of contaminant which, based on simulation results, can be very influential in contaminant migration. Sample results supported previous observations of empty to partially empty pesticide containers buried in the trenches and the primary contaminant was disulfoton. While field investigations conducted as part of the RI could not precisely determine the quantity of contaminant, observations put the amount closer to the minimum amount of product originally estimated (WASTREN, 1993).

## SUMMARY OF MODEL RESULTS

Because of the complexity of site conditions and contaminant properties, a combination of three models were applied to simulate the fate and transport of disulfoton and its metabolites at the Murtaugh landfill, which included detailed water balance simulation and transport of aqueous-phase and non-aqueous phase liquid (NAPL) flow of disulfoton. Results of the organic contaminant transport and fate study are summarized as follows.

1. Only slight percolation was predicted at the Murtaugh site because of the low precipitation and relatively large potential evapotranspiration.
2. Volume and concentration of disulfoton are major factors in chemical migration. Large volumes of liquid chemical will create tremendous gravitational potential for downward movement; high concentrations create a non-aqueous phase liquid (NAPL) disulfoton.
3. The low permeable caliche layer stops NAPL flow and deters downward movement of dissolved chemicals. Without the caliche layer, however, both the NAPL and dissolved chemical can reach the fractured basalt very easily. Once the chemicals reach the fractured basalt, they could travel to the deeper geological materials or the water table by preferential flow path, such as interconnected fractures and faults.
4. When primary degradation is simulated, disulfoton disappears a few years after leaking. However, fate of toxic metabolites remains uncertain.

While the caliche was shown to significantly deter contaminant migration, neither the caliche nor dense basalt layers were proven to be continuous beneath the site. Limited data from boreholes and wells in the area indicate that the caliche extends beneath the disposal trenches. If both layers were known to be continuous, it could be stated with more certainty that contaminants will not reach the groundwater. However, given the low solubility of disulfoton and the evidence that the actual source term is close to the minimum estimates rather than the maximum (as simulated in scenario 3), downward migration of disulfoton to the ground water was felt to be highly unlikely.

## RISK ASSESSMENT

Both human health and environmental risk assessment were conducted for the Murtaugh landfill as part of the RI (WASTREN, 1993). Based on the fate and transport modeling, groundwater health standards were not likely to be exceeded under a "No Action" remediation option within the 100-year simulation period. Because surface soil and air samples were not found to be contaminated, inhalation of contaminants (airborne, dust or vapors) or exposure to surface soils did not pose undue risk. Thus no direct actions appeared necessary to reduce these risks to acceptable levels. Based on the human health and environmental risk assessment, the greatest threat was subsurface exposure to burrowing rodents and other wildlife populations.

## SITE REMEDIATION

The purpose of any remedial action was to eliminate or reduce to acceptable levels the potential for direct contact, inhalation or ingestion by human or environmental receptors. Thus, in remediation of the site it was necessary to address unacceptably high risk factors and ensure that the existing situation was not aggravated. Specifically, any remediation measure must not increase the potential for contaminant migration to the groundwater or contamination of the soil surface air. Based on the RI, Knowles (1993) examined eight alternatives for remediation of the site: (1) no action; (2) access restriction; (3) capping; (4) thermal desorption; (5) hydrolysis; (6) land farming; (7) incineration; and (8) off-site treatment and disposal. Due to the large number of buried containers compared to the small amount of pesticide actually observed in the RI, results from the modeling indicating no violation of groundwater health standards, the fact that no hot spots were found, and the limited potential risk presented by the site, removal and treatment of the contaminant was not deemed cost effective. Model results indicating only slight percolation at the site indicated that capping the site with an impermeable layer was not necessary.

With the preceding conditions in mind, closure of the site included access restriction and covering the site with 2 ft (0.6 m) of fill material to reduce the likelihood of subsurface exposure to burrowing rodents. This addresses the significant potential risks of contact with contaminated subsurface soils and exposure to buried containers. This additionally adds a buffer zone against percolation of water to the contaminant. Access wells installed to sample contamination within the unsaturated zone beneath the trenches will be monitored for contaminant migration. Additionally, the site will remain the possession of the BLM who will maintain the site to limit access and to ensure that future development or use of the site will not compromise human or environmental health and safety. Because this remedy results in hazardous substances remaining on-site above health-based levels, a review will be conducted by the BLM within five years after commencement of remedial action to ensure that the remediation continues to provide adequate protection for human and environmental health.

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