VADOFT allows consideration of multiple soil layers, each with homogeneous soil characteristics. Within the unsaturated zone, the attenuation of organic pollutants is predicted based on longitudinal dispersion, an estimated retardation coefficient derived from an equilibrium partition coefficient, and a first-order rate of pollutant degradation. The input requirements for the unsaturated zone module include various site-specific and geologic parameters and the rate of groundwater recharge in the area of the site. It is assumed that the flux of pollutant mass into the unsaturated zone beneath a land application site can be represented by results from the mass balance calculations described above. Results from analysis of the unsaturated zone give the flow velocity and concentration profiles for each pollutant of interest. These velocities and concentrations are evaluated at the water table, converted to a mass flux, and used as input to the AT123D saturated zone module.

The flow system in the vertical column is solved with VADOFT, that is based on an overlapping representation of the unsaturated and saturated zones. The water flux into the unsaturated zone is specified for the bottom of the zone of incorporation for sewage sludge. In addition, a constant pressure-head boundary condition is specified for the bottom of the unsaturated zone beneath the land application site. This pressure-head is chosen to be consistent with the expected pressure head at the bottom of the saturated zone. Transport in the unsaturated zone is determined using the Darcy velocity and saturation profiles from the flow simulation. From these, the transport velocity profile can be determined.

Although limited to one-dimensional flow and transport, the use of a rigorous finite-element model in the unsaturated zone allows consideration of depth-variant physical and chemical processes that would influence the mass flux entering the saturated zone. Among the more important of these processes are advection (that is a function of the Darcy velocity, saturation and porosity), mass dispersion, adsorption of the leachate onto the solid phase, and both chemical and biological degradation.

To represent the variably saturated soil column beneath the application site, the model discretizes the column into a finite-element grid consisting of a series of one-dimensional elements connected at nodal points. Elements can be assigned different properties for the simulation of flow in a heterogenous system. The model generates the grid from user-defined zones; the user defines the homogeneous properties of each zone, the zone thickness and the number of elements per zone, and the code automatically divides each zone into a series of elements of equal length. The governing equation is approximated using the Galerkin finite element method and then solved iteratively for the dependent variable (pressure-head) subject to the chosen initial and boundary conditions. Solution of the series of nonlinear simultaneous equations generated by the Galerkin scheme is accomplished by either Picard iteration, a Newton-Raphson algorithm or a modified Newton-Raphson algorithm. Once the finite-element calculation converges, the model yields estimated values for all the variables at each of the discrete nodal points. A detailed description of the solution scheme is found in U.S. EPA (1989g).

One-dimensional, advective-dispersive transport is estimated with VADOFT based on the estimated mass flux of pollutant into the top of the soil column, and a zero concentration boundary condition at the bottom of the saturated zone. The resulting mass flux from the VADOFT simulation is used as input for the AT123D model, that simulates pollutant transport

through the saturated zone. It is represented as a mass flux boundary condition applied over a rectangular area representative of the land application site. The transient nature of the flux into the saturated zone is represented by time-dependent levels interpolated from the results generated by the VADOFT simulation.

As in calculations for the unsaturated zone, degradation of organic pollutants is assumed to be first-order during transport through the aquifer. Speciation and complexation reactions are ignored for metals, leading to the possible over- or underestimation of expected concentrations of metals in groundwater at the location of a receptor well. Detailed descriptions of the AT123D model are provided by U.S. EPA (1986d) and by Yeh (1981) and will not be repeated here. In general, the model provides an analytical solution to the basic advective-dispersive transport equation. One advantage of AT123D is its flexibility: the model allows the user up to 450 options and is capable of simulating a wide variety of configurations of source release and boundary conditions. For the current application, AT123D uses the source term from VADOFT and other input parameters to predict concentrations of pollutant within 300 years in a receptor well at the downgradient edge of the land application site.

Estimating Human Exposure

Once pollutant concentrations in groundwater are estimated, estimates of human exposure are made based on assumptions about the rate that the average individual consumes drinking water. Potential exposure through ingestion of contaminated groundwater is estimated as:

$$EXP_j = \frac{C_{wel,j} \cdot IW}{BW} \tag{4-77}$$

where:

EXP_j	=	exposure to pollutant j in sewage sludge (mg pollutant/kg body weight-day),
$C_{wel,j}$	=	concentration of pollutant j in well water (mg pollutant/L water),
IW	=	volume of water ingested daily (L water/day), and
BW	=	body weight (kg).

Data Inputs

All the data inputs required for this pathway are presented in Section 4.2.12.

Example Exposure Calculations for Pathway 14

The following example calculates exposure of humans to dioxins and dibenzofurans through ingestion of groundwater that has received leachate from sewage sludge-amended soil on agricultural land. The mass balance portion of the calculation, which is the same for Pathways 12, 13, and 14, is presented in the section "Example Exposure Calculations for Pathway 12". In the mass balance calculation, f_{lec} is estimated from Eq. 4-27:

$$f_{lec} = \frac{1.8x10^{-4}/yr}{2.3x10^{-2}/yr} = 7.8x10^{-3}$$

where:

1.8×10^{-4}	=	K_{lec} (loss rate due to leaching of dioxins from sewage sludge-amended land) calculated in Pathway 12, and
2.3×10^{-2}	=	K_{tot} (total loss rate for dioxins from sewage sludge-amended land) calculated in Pathway 12.

The duration of the square wave for approximating the loading of dioxins into the unsaturated soil zone is calculated using Eq. 4-74:

$$TP = \frac{20yr}{[1 - e^{(-2.3x10^{-2}yr^{-1} \cdot 20yr)}]} = 53yr$$

where:

20	=	N (total number of years sewage sludge is applied to land) from Exhibit 4-3, and
$2.3x10^{-2}$	=	K_{tot} (total loss rate for dioxins from sewage sludge-amended land) calculated in Pathway 12.

Eq. 4-75 is then used to calculate the average flux of pollutant leaching from the land application site:

$$FA_{lec,dioxins} = \frac{20yr \cdot 7Mg/ha\text{-}yr \cdot 3.11x10^{-4}mg/kg \cdot 7.8x10^{-3} \cdot 0.001}{53yr}$$
$$= 6.4x10^{-9}kg/ha\text{-}yr$$

where:

20	=	N (total number of years sewage sludge is applied to land) from Exhibit 4-3,
7	=	AR (annual whole sludge application rate of sewage sludge to land) from Exhibit 4-3,
$3.11x10^{-4}$	=	C_j (concentration of dioxins in sewage sludge) from Exhibit 4-1,
$7.8x10^{-3}$	=	f_{lec} (fraction of total pollutant loss caused by leaching) calculated above,
0.001	=	constant to convert (Mg-mg/kg) to (kg), and
53	=	TP (duration of "square" wave for approximating the loading of pollutant into the unsaturated soil zone) calculated above.

Eq. 4-76 is then used to calculate the average concentration of pollutant in the leachate:

$$C_{lec,dioxins} = \frac{0.1 \cdot 6.4x10^{-9} kg/ha\text{-}yr}{0.5 m/yr}$$
$$= 1.3x10^{-9} mg/L$$

where:

0.1	=	constant to convert units from (kg/ha-m) to (mg/L),
$6.4x10^{-9}$	=	$FA_{lec,\,j}$ (annual flux of dioxins leaching from the site) calculated above, and
0.5	=	*NR* (net recharge to groundwater in treated area) from Exhibit 4-16.

The concentration of dioxins in leachate is then converted into a well concentration through the use of VADOFT and AT123D. The concentration of dioxins in the well is estimated to be 0 mg/L; i.e., dioxins are not transported through the unsaturated and saturated zones to the well in appreciable concentrations. Using Eq. 4-77, human exposure to dioxins in groundwater is thus estimated to be:

$$EXP_{dioxins} = \frac{0 mg/L \cdot 2 L/day}{70 kg} = 0 mg/kg\text{-}day$$

where:

0	=	$C_{wel,\,j}$ (concentration of dioxins in well water) obtained through VADOFT and AT123D modeling,
2	=	*IW* (quantity of water ingested daily) from Exhibit 4-16, and
70	=	*BW* (body weight), assumed to be 70 kg.

4.2.15 Pathway 15 - Infant Exposure to Pollutants Through Breastfeeding

Pathway 15 evaluates exposures of infants to pollutants in breast milk. Only highly lipophilic pollutants are evaluated because these are expected to concentrate in milkfat. Concentrations of pollutants in breast milk result in part from the mother's exposure to pollutants in sewage sludge through several human exposure pathways, including pathways 1, 2, 4, 5, 12, 13 and 14, as described in previous sections. In addition, the mother is assumed to be exposed to background concentrations of pollutants from sources other than sewage sludge. This analysis presents two exposure scenarios that differ in exposure duration and/or percent of an infant's lifetime over which dose is averaged.

Methods

The method used to estimate infant exposure is taken from *Estimating Exposures to Dioxin-Like Compounds* (U.S. EPA, 1994b), and is based on an approach developed by Smith (1987). The method assumes that the concentration of pollutant in breast milk fat is the same as the concentration in maternal fat. The following calculation is used:

$$C_{milkfat,\,j} = \frac{(m_{background,\,j} + m_{sludge,\,j}) \cdot h_j \cdot f_{1,\,j}}{0.693 \cdot f_2} \qquad (4\text{-}78)$$

where:

$C_{milkfat,\,j}$	=	concentration of pollutant j in maternal milk (mg pollutant/kg milk fat),
$m_{background,\,j}$	=	maternal intake of pollutant j from sources other than sewage sludge (mg pollutant/kg body weight-day),
$m_{sludge,\,j}$	=	maternal intake of pollutant j from relevant sewage sludge exposure pathways (mg pollutant/kg body weight-day),
h_j	=	half-life of pollutant j in adults (days),
$f_{1,\,j}$	=	proportion of ingested pollutant j that is stored in fat (dimensionless),
0.693	=	ln(0.5), to convert the half-life of pollutant j into a rate (dimensionless), and
f_2	=	proportion of mother's weight that is fat (kg maternal fat/kg total body weight).

This steady-state model assumes that the pollutant levels in maternal fat remain constant, i.e., changes in maternal levels of pollutant during breastfeeding do not occur.

The concentration of pollutant in mother's milk is then used in the following equation to estimate the daily dose to the infant:

$$EXPI_j = \frac{C_{milkfat,\,j} \cdot f_3 \cdot f_{4,\,j} \cdot IM \cdot ED}{BW_{infant} \cdot AT} \qquad (4\text{-}79)$$

where:

$EXPI_j$	=	infant's average daily exposure to pollutant j (mg pollutant/kg body weight-day),
f_3	=	proportion of fat in breast milk (dimensionless),
$f_{4,\,j}$	=	proportion of ingested pollutant j that is absorbed (dimensionless),
IM	=	ingestion rate of breast milk (kg milk/day),
ED	=	exposure duration (yr),

BW_{infant}	=	average body weight of the infant during the exposure period (kg), and
AT	=	averaging time (yr).

Daily doses to the infant are determined for each of the two following scenarios, which are distinguished by exposure duration and averaging time:

Scenario 1: An infant breastfeeds for one year and the daily dose is averaged over this one year exposure period. Infant body weight at one-half exposure duration (six months) is used to represent the average weight over the exposure duration in this scenario.

Scenario 2: Under the second scenario, the infant breastfeeds for two years. The dose is averaged over 70 years.

The data used in these scenarios are discussed in the next section.

Data Inputs

The exposure scenarios were constructed using a mix of central tendency and high-end values. The central tendency values were used for the following parameters: background maternal intake, proportion of pollutant stored in fat, fat content of breast milk, and absorption rate for ingested pollutants. The remainder of the parameters were set to their high-end values. Sources for the input values are discussed below.

General Inputs. Exhibit 4-19 shows the general (i.e., not pollutant-specific) inputs used to calculate exposure to infants through breastfeeding. The values for many of the input parameters were taken from Smith (1987) and U.S. EPA (1994b). The fat content of milk (0.04) and the ingestion rate of milk come from information reported in Smith (1987). A study of British children found mean intakes in the first 7 to 8 months that ranged from 677 to 922 ml per day (Whitehead and Paul, 1981 as cited in Smith, 1987). A value of 0.9 L is used in this assessment; assuming the density of breast milk is approximately that of water, 0.9 L is equivalent to 0.9 kg. Smith (1987) presents two studies that estimate the percent of maternal body weight that is fat. One study estimated a fat content of about 33% (Timson and Coffman, 1984, as cited in Smith, 1987), and a second study found a reduction from a mean of 28% to 26.3% during four months of lactation (Butte et al., 1984, as cited in Smith, 1987). Based on these data, the current assessment uses 0.3 as the proportion of maternal weight that is fat.

The exposure duration and infant body weight over the exposure period differ between the two scenarios. This analysis uses exposure durations of one or two years. Because the body weight of an infant differs for these two exposure durations, the current analysis uses different infant body weights for each exposure duration. For a one-year exposure duration, an infant body weight of 9.1 kg (an average for babies 6 to 11 months old) is used to represent an average body weight during the first year. For the two year exposure duration, a body weight of 11.3 kg (the average for 1 year old babies) is used as the average body weight over the first two years

of life. Both values are taken from the National Center for Health Statistics (1987) (cited in U.S. EPA, 1994b).

The appropriate choice of averaging time for these less-than-lifetime exposures depends on the health endpoint assessed. In this analysis, different averaging times are used to estimate the daily dose. First, the averaging time is set equal to the exposure duration (Scenario 1) to obtain an estimate of daily dose for the period during which exposure occurs. Such an estimate may be appropriate to evaluate health effects (such as developmental effects) that can occur from short-term exposures (U.S. EPA, 1994b). (At this time, corresponding health risk values are not available. Therefore, this calculation is only carried to the point of estimating exposure.) Second, this analysis also uses an averaging time of 70 years (Scenario 2) to calculate a lifetime average daily dose (LADD) to correspond with exposure duration assumed for cancer potency estimates.

EXHIBIT 4-19
General Input Parameters to Estimate Exposure to Pollutants through Breastfeeding

Parameter		Value
Parameter values common to all three scenarios:		
f_2: proportion of mother's weight that is fat (dimensionless)		0.3
f_3: proportion of fat in breast milk (dimensionless)		0.04
IM: ingestion rate of breast milk (kg/day)		0.9
Parameter values that differ among scenarios:		
ED: exposure duration (yr)	scenario 1 scenario 2	1 2
BW_{infant}: infant's average body weight (kg)	scenario 1 scenario 2	9.1 11.3
AT: averaging time (yr)	scenario 1 scenario 2	1 70

Pollutant-Specific Inputs. For this pathway, only pollutants with a log K_{ow} value greater than five were evaluated. Other pollutants were not considered sufficiently lipophilic to warrant further analysis. Of the Round Two candidate pollutants, only three--dioxins and dibenzofurans, coplanar PCBs and bis (2-ethylhexyl) phthalate (BEHP)--have log K_{ow} values of five or greater (see Exhibit 4-17 in Section 4.2.12). After further research, infant exposure to BEHP through breastfeeding was not evaluated because the biological half-life of BEHP in humans is less than one day. Schmid and Schlatter (1985) estimated the urinary elimination half-life to be 12 hours and concluded that accumulation of BEHP in the human body was unlikely. Sjoberg et al.

(1985) determined that BEHP levels in the human body decline with a half-life of 10 hours. Based on these data, infant exposure to BEHP through breastfeeding did not require further consideration. Therefore, only dioxins and furans and coplanar PCBs are considered for this pathway.

There are five pollutant-specific parameters used to estimate exposure through this pathway: maternal background exposure from other sources; maternal exposure from the relevant sewage sludge pathways; percentage of pollutant ingested by the mother that is stored in her fat; the half-life of the pollutant in the adult human body; and the percentage of pollutant ingested by the infant that is absorbed. The pollutant-specific inputs for these pollutants are described below; Exhibit 4-17 summarizes the pollutant-specific input values and the references for these values.

Maternal exposure from other sources. For dioxins, a background exposure of 119 pg/day is assumed. This value is derived from environmental concentration data collected in rural, pristine, and urban areas not thought to be affected by local sources (U.S. EPA, 1994b). Dividing this value by the standard adult body weight of 70 kg yields 1.7 pg/kg-day. A similar value for coplanar PCBs was not found; therefore, only incremental risk from exposure above background can be assessed for coplanar PCBs.

Maternal exposure from sewage sludge pathways. For the estimate of maternal exposure from the relevant sewage sludge pathways, the results of the exposure assessments for pathways 1, 2, 4, 5, 12, 13 and 14 (described in previous sections) were considered simultaneously as the source of maternal body burden.

Percentage of pollutant ingested by the mother stored in her fat. The proportion of ingested dioxins stored in fat (0.9) is taken from Smith (1987) (as cited in U.S. EPA, 1994b). Coplanar PCBs are assumed to behave in a similar manner. Distribution studies of PCBs demonstrate that the adipose/plasma partition ratio for PCBs ranged between 185/1 and 210/1 depending on the PCB involved (Brown and Lawton, 1984). Based on these data (and assuming some PCBs are stored in other body tissues besides adipose tissues and plasma), it is assumed that 90 percent of the ingested coplanar PCBs are stored in fat.

Biological half-lives. U.S. EPA (1994b) presents different half-lives for 2,3,7,8-TCDD in humans, from 5.8 years to 7 years. For this analysis, a half-life of 7 years is used as a conservative estimate.

For PCBs, Yakushiji et al. (1978) reported an approximate half-life of 8 months. In an analysis of exposure to fish contaminated by PCBs in the Great Lakes, Anderson and Amrhein (1993) assumed a half-life of one year, based on their review of the literature, although it should be noted that longer half-lives were estimated for particular PCB mixtures (up to a suggested half-life of 10 years for congener #153). For this assessment, a half-life of one year was used for coplanar PCBs.

Percentage of pollutant ingested by the infant that is absorbed. Ninety percent of ingested dioxins were assumed to be absorbed by the infant (Smith, 1987, as cited in U.S. EPA, 1994b). For PCBs, studies of monkeys (Allen et al., 1974) and ferrets (Bleavins et al., 1984) demonstrated 90% and 85.4% absorption efficiencies, respectively. Based on the data for monkeys, 90% absorption efficiency in humans was assumed for coplanar PCBs.

EXHIBIT 4-20
Pollutant-Specific Input Parameters to Estimate Exposure to Pollutants through Breastfeeding

Pollutant	Maternal Background Intake (mg/kg-day) ($m_{background,\ j}$)	Proportion of Pollutant Stored in Fat ($f_{1,\ j}$)	Proportion of Ingested Pollutant Absorbed ($f_{4,\ j}$)	Half-Life of Pollutant in Adults (h_j)
Dioxins and Dibenzofurans	1.7×10^{-9} [1]	0.9[2]	0.9[2]	7 years[1]
Polychlorinated biphenyls (coplanar)		0.9[3]	0.9[4]	1 year[5]

[1]U.S. EPA (1994b).
[2]Smith (1987).
[3]Brown and Lawton (1984).
[4]Allen et al. (1974).
[5]Anderson and Amrhein (1993).

Example Exposure Calculations for Pathway 15

The following example calculates the exposure of infants to dioxins and dibenzofurans in breast milk. First, the maternal intake of dioxins in sewage sludge from agricultural pathways 1, 2, 4, 5, 12, 13, and 14 ($m_{sludge,\ dioxins}$) is calculated using the methods discussed in the corresponding sections. The value of $m_{sludge,\ dioxins}$ is 1.4×10^{-9} mg/kg-day. Equation 4-78 is then used to calculate the concentration of dioxins in milk fat:

$$C_{milkfat,dioxins} = \frac{(1.7x10^{-9} mg/kg\text{-}day + 1.4x10^{-9} mg/kg\text{-}day) \cdot 2555 days \cdot 0.9}{0.693 \cdot 0.3}$$

$$= 3.5x10^{-5} mg/kg$$

where:

$1.7x10^{-9}$	=	$m_{background,\ j}$ (maternal intake of dioxins from sources other than sewage sludge) from Exhibit 4-20,
$1.4x10^{-9}$	=	$m_{sludge,\ j}$ (maternal intake of dioxins from sewage sludge exposure pathways 1, 2, 4, 5, 12, 13, and 14),
2555	=	h_j (half-life of dioxins in adults) from Exhibit 4-20,
0.9	=	$f_{1,\ j}$ (proportion of ingested dioxins that is stored in fat) from Exhibit 4-20,
0.693	=	ln(0.5), to convert the half-life of dioxins into a rate, and
0.3	=	f_2 (proportion of mother's weight that is fat) from Exhibit 4-19.

Exposure to the infant under Scenario 2 can then be calculated using Eq. 4-79:

$$EXPI_{dioxins} = \frac{3.5x10^{-5} mg/kg \cdot 0.04 \cdot 0.9 \cdot 0.9 kg/day \cdot 2 yr}{11.3 kg \cdot 70 yr}$$

$$= 2.8x10^{-9} mg/kg\text{-}day$$

where:

$3.5x10^{-5}$	=	$C_{milkfat,\ dioxins}$ (concentration of dioxins in maternal milk) calculated above,
0.04	=	f_3 (proportion of fat in breast milk) from Exhibit 4-19,
0.9	=	$f_{4,\ j}$ (proportion of ingested dioxins that is absorbed) from Exhibit 4-20,
0.9	=	IM (ingestion rate of breast milk) from Exhibit 4-19,
2	=	ED (exposure duration) from Exhibit 4-19 for Scenario 2,
11.3	=	BW_{infant} (average body weight of the infant during the exposure period) from Exhibit 4-19 for Scenario 2, and
70	=	AT (averaging time) from Exhibit 4-19 for Scenario 2.

4.3 SURFACE DISPOSAL EXPOSURE METHODOLOGIES

Under the surface disposal practice for sewage sludge, humans can be exposed to pollutants volatilizing or leaching from sewage sludge placed in either a monofill or a surface impoundment. Humans can either inhale air or ingest groundwater containing pollutants. In this section, the modeling prototypes for monofills and surface impoundments are defined. The methods used to model volatilization and leaching from the prototypes are then presented.

4.3.1 Definitions of a Monofill and a Surface Impoundment

The monofill prototype is a sewage sludge-only trench fill. Only de-watered sewage sludges with solids content greater than or equal to 20 percent are assumed to be suitable for placement in a monofill, and the sewage sludge often is mixed with a bulking agent (e.g., soil) to increase solids content. Operating procedures assumed include daily cover which reduces odors and provides vector control, and a final cover placed on the monofill after closure.

The surface impoundment prototype receives a continuous inflow of sewage sludge with a low solids content (between two and five percent). A vertical outflow pipe maintains the surface liquid level at a constant height, and liquid is assumed to leave the impoundment both in the outflow (possibly for return to the treatment works) and in seepage through the floor of the impoundment. Over time, particulate settling occurs and a denser layer of solids accumulates on the floor of the impoundment. Eventually, this layer of solids reaches the top of the impoundment and no further inflow is possible. Upon closure, the sewage sludge is left permanently in place and remains uncovered.

One key difference between the surface impoundment and monofill prototypes is that the active surface impoundment is assumed to contain significantly more liquid than the active monofill. Seepage through the floor of the surface impoundment is expected to be greater than seepage from the monofill, and may be sufficient to sustain a local mounding of the underlying water table. The surface layer of the impoundment also is assumed to be in a liquid state over the active lifetime of the impoundment. The volatilization of organic pollutants from this liquid layer is expected to differ from that predicted for a monofill, which is assumed to contain a higher percentage of solids and to receive a daily, and eventually a permanent, cover.

4.3.2 Methods for the Monofill Prototype

Method for Mass Balance

Pollutant mass is assumed to enter the monofill through daily deposits of sewage sludge and to be removed through degradation, leaching, and volatilization. Rates of pollutant loss are assumed to be first-order (that is, proportional to the residual concentration of pollutant in the monofill); mass balance calculations begin by estimating first-order loss coefficients for each competing loss process.

Pollutant Losses Through Leaching. A coefficient for the rate of pollutant loss to leaching is calculated by assuming that pollutant mass in a filled monofill active sewage sludge unit is partitioned at equilibrium between dissolved and adsorbed phases. Based on mathematical relationships presented in Section 4.2.14, the concentration of pollutant dissolved in water within the monofill can be estimated from the total concentration of pollutant within the monofill:

$$C_w = \frac{C_t}{\left[BD_{mix} \cdot K_d \cdot 10^{-3} + \theta_w + \dot{H}\,\theta_a\right]} \tag{4-80}$$

where:

C_w	=	concentration of dissolved pollutant in water-filled pore space of sewage sludge/soil in monofill (kg pollutant/m^3 porewater),
C_t	=	total concentration of pollutant in bulk sewage sludge/soil in monofill (kg pollutant/m^3 total bulk soil volume),
BD_{mix}	=	bulk density of sewage sludge/soil (kg soil/m^3 total bulk soil volume),
K_d	=	soil-water partition coefficient (L water/kg soil),
θ_w	=	water-filled porosity (dimensionless),
$\dot{H}$	=	Henry's Law constant (dimensionless),
θ_a	=	air-filled porosity (dimensionless), and
10^{-3}	=	constant to convert (L) to (m^3).

The dimensionless Henry's Law constant can be calculated by:

$$\dot{H} = \frac{H}{R \cdot T \cdot 10^{-3}} \tag{4-81}$$

where:

H	=	Henry's Law constant (atm-m^3/mol),
R	=	gas constant (L-atm/mol-K),
T	=	temperature (Kelvin), and
10^{-3}	=	constant to convert (L) to (m^3).

For an arbitrary unit concentration of pollutant in the sewage sludge/soil (1 kg pollutant/m^3 sewage sludge/soil), a flux of pollutant mass leaching from the monofill can be calculated as the product of net recharge (*NR*) and the expected concentration of pollutant in leachate. Moreover, with a unit concentration of pollutant, the mass of pollutant beneath one square meter of surface is equal to the volume of sewage sludge/soil beneath that area. This volume can be expressed as kg pollutant/m^2 area/m depth of the monofill. As discussed in Section 4.2.14, the estimated flux of leaching pollutant is divided by this mass to derive a first-order loss coefficient for leaching:

$$K_{lec} = \frac{NR}{\left(BD_{mix} K_d \cdot 10^{-3} + \theta_w + \dot{H}\,\theta_a\right) d_{mf}} \tag{4-82}$$

where:

K_{lec}	=	loss rate due to leaching of the pollutant from monofill (yr^{-1}),
NR	=	annual recharge to groundwater beneath the monofill (m recharge/yr), and
d_{mf}	=	depth of a monofill active sewage sludge unit (m).

Pollutant Losses Through Volatilization. Rates of volatilization from a filled active sewage sludge unit in a monofill will vary according to whether a cover layer of soil has been applied. Each active sewage sludge unit in the monofill is assumed to contain uncovered sewage sludge for a few hours on each of the days it receives sewage sludge. Following each placement, a temporary cover layer of soil is applied. Once the monofill's capacity is exhausted, a thicker permanent cover of soil is applied to the entire monofill (U.S. EPA, 1986d). A time-weighted average of emission rates with and without cover is used, therefore, to describe the average rate of volatile emissions for an individual active sewage sludge unit in the monofill. The fraction of the monofill's active lifetime that a typical active sewage sludge unit will be uncovered is calculated as:

$$f_{un} = \frac{t_{un}}{LF} \tag{4-83}$$

where:

f_{un}	=	fraction of monofill's active lifetime that a typical active sewage sludge unit contains sewage sludge without soil cover (dimensionless),
t_{un}	=	time that a typical active sewage sludge unit contains uncovered sewage sludge (yr), and
LF	=	active lifetime of monofill (yr).

Some monofill active sewage sludge units will be filled early in the monofill's operation, others nearer to the monofill's closure. The average monofill active sewage sludge unit is assumed to contain sewage sludge for half the active lifetime of the monofill. The fraction of the monofill's active lifetime that such a unit will contain sewage sludge that is covered is:

$$f_{co} = \frac{1}{2} - f_{un} \tag{4-84}$$

where:

f_{co}	=	fraction of monofill's active lifetime that typical active sewage sludge unit contains sewage sludge with temporary soil cover (dimensionless), and
½	=	fraction of monofill's active lifetime that typical active sewage sludge unit contains sewage sludge (dimensionless).

A time-weighted average flux of pollutant emissions from a typical monofill active sewage sludge unit is calculated from equations describing emissions from a unit with and without soil cover. According to Environmental Science and Engineering (1985) as discussed in U.S. EPA (1986d), emissions from an uncovered landfill cell can be described by:

$$q_{un} = \frac{0.17\ u\ 0.994^{(T-293)}\ C_a}{\sqrt{MW}} \tag{4-85}$$

where:

q_{un}	=	pollutant flux from sewage sludge/soil for uncovered period (kg pollutant/m^2 unit-sec),
u	=	average wind speed (m/sec),
T	=	temperature (Kelvin),
C_a	=	concentration of gaseous pollutant in air-filled pore space of sewage sludge/soil (kg pollutant/m^3 air), and
MW	=	molecular weight of pollutant (g/mol).

For a sewage sludge unit with soil cover:

$$q_{co} = \frac{9.2\times10^{-5}\ \theta_{ca}^{(10/3)}\ 1.006^{(T-293)}\ C_a}{\sqrt{MW}\ d_c\ \theta_{ct}^2} \tag{4-86}$$

where:

q_{co}	=	pollutant flux from treated soil for covered period (kg/m^2-sec),
θ_{ca}	=	air-filled porosity of cover layer of soil (dimensionless),
θ_{ct}	=	total porosity of cover layer of soil (dimensionless), and
d_c	=	depth of soil cover (m).

Equations 4-85 and 4-86 require an estimate of the concentration of pollutant in air-filled pore space within the active sewage sludge unit. As discussed in Section 4.2.14, this concentration can be related to the total concentration of pollutant in sewage sludge/soil as:

$$C_a = \frac{C_t}{\frac{K_d\ BD_{mix}\ 10^{-3}}{\dot{H}} + \frac{\theta_w}{\dot{H}} + \theta_a} \tag{4-87}$$

where:

C_a	=	concentration of gaseous pollutant in air-filled pore space of monofill (kg pollutant/m^3 air),
C_t	=	total concentration of pollutant in bulk sewage sludge/soil in monofill (kg pollutant/m^3 total bulk soil volume),
K_d	=	soil-water partition coefficient (L water/kg soil),
BD_{mix}	=	bulk density of sewage sludge/soil (kg soil/m^3 total bulk soil volume),
$\dot{H}$	=	Henry's Law constant (dimensionless),
θ_w	=	water-filled porosity (dimensionless),
θ_a	=	air-filled porosity (dimensionless), and
10^{-3}	=	constant to convert (L) to (m^3).

Estimated pollutant fluxes from an uncovered and temporarily covered monofill active sewage sludge unit are combined to derive a time-weighted average pollutant flux from a monofill unit during the monofill's active lifetime:

$$q_{ac} = q_{un}\, f_{un} + q_{co}\, f_{co} \tag{4-88}$$

where:

q_{ac}	=	time-weighted average pollutant flux from typical monofill unit over the active lifetime of the monofill (kg pollutant/m^2 unit -sec).

For a unit concentration (C_t=1 kg pollutant/m^3 sewage sludge/soil) of pollutant in sewage sludge/soil, the mass of pollutant beneath one square meter of monofill surface (kg/m^2) is equal to the depth of the monofill (m). Therefore, converting the estimated loss rate (kg/m^2-sec) into a first-order loss coefficient (yr^{-1}) requires division by depth and adjustment of units from seconds to years:

$$K_{va} = \frac{q_{ac}\ 3.16\times10^{7}}{d_{mf}} \tag{4-89}$$

where:

K_{va}	=	loss rate of pollutant due to volatilization during monofill's active operation (yr^{-1}), and
$3.16\text{x}10^{7}$	=	constant to convert units from (sec^{-1}) to (yr^{-1}).

Fraction of Pollutant Loss to Each Pathway. Estimated coefficients for losses to volatilization and leaching are combined with assumed rates of degradation to yield a "lumped" coefficient describing pollutant loss through all three pathways during the monofill's active lifetime:

$$K_{ta} = K_{lec} + K_{va} + K_{deg} \tag{4-90}$$

where:

K_{ta}	=	total loss rate of pollutant due to leaching, volatilization, and degradation during monofill's active operation (yr^{-1}).

The fraction of pollutant loss attributable to each individual process during the monofill's active lifetime is:

$$f_{la} = \frac{K_{lec}}{K_{ta}} \qquad f_{va} = \frac{K_{va}}{K_{ta}} \qquad f_{da} = \frac{K_{deg}}{K_{ta}} \tag{4-91}$$

where:

f_{la}	=	fraction of total pollutant loss during monofill's active operation attributable to leaching (dimensionless),
f_{va}	=	fraction of total pollutant loss during monofill's active operation attributable to volatilization (dimensionless), and
f_{da}	=	fraction of total pollutant loss during monofill's active operation attributable to degradation (dimensionless).

The fraction of total loading lost within the monofill's active lifetime is calculated numerically from the lumped rate of pollutant loss, assuming a time step of one year and a unit pollutant loading of one kg/ha-yr:

$$M_t = 0 \qquad (t=0)$$
$$M_t = [M_{t-1} + 1]\, e^{-K_{ta}} \qquad (1 \le t \le LF) \tag{4-92}$$

The fraction of total pollutant lost during a monofill's active lifetime can then be calculated as:

$$f_{ac} = \frac{1 \cdot LF - M_{LF}}{1 \cdot LF} \tag{4-93}$$

where:

f_{ac}	=	fraction of total pollutant lost during monofill's active lifetime (dimensionless),
M_{LF}	=	mass of pollutant in sewage sludge/soil at end of monofill's active lifetime (kg pollutant/ha), and
1	=	annual unit loading of pollutant (kg/ha-yr).

Once the monofill's capacity is exhausted, a permanent cover layer of soil is applied to its surface. This permanent cover reduces the rate of volatilization, changing both the total rate of pollutant loss and the relative fraction of that loss attributable to volatilization, leaching, or degradation. Based on the increased thickness of cover, an estimated rate of volatilization from the inactive monofill (K_{vi}) is calculated with Eqs. 4-86 through 4-88 by setting f_{un} to zero. Rate coefficients for loss to leaching and degradation are assumed to be unaffected by soil cover, so the lumped rate of loss for the inactive monofill is described by:

$$K_{ti} = K_{lec} + K_{vi} + K_{deg} \tag{4-94}$$

where:

K_{ti}	=	total loss rate of pollutant from inactive monofill (yr^{-1}), and
K_{vi}	=	loss rate of pollutant due to volatilization from inactive monofill (yr^{-1}).

The fraction of loss attributable to volatilization is calculated as:

$$f_{vi} = \frac{K_{vi}}{K_{ti}} \tag{4-95}$$

where:

f_{vi}	=	fraction of total pollutant loss from inactive monofill attributable to volatilization (dimensionless).

As will be discussed below, these fractions and the lumped rate coefficients for pollutant loss are used to estimate pollutant concentrations in air and groundwater near the site.

Method for Groundwater Pathway

Upon completion of the mass balance calculations described above, two additional steps are used to calculate the concentration of each pollutant in groundwater:

1) Determine the concentration of pollutant in leachate from the bottom of the monofill.

2) Use mathematical models for the transport of pollutant through the unsaturated and saturated soil zones to estimate expected concentrations of pollutants in groundwater.

With the mass balance calculations, the total rate at which a pollutant is lost from the monofill, and the fraction of that loss attributable to leaching, are estimated. The amount of time that would be required to deplete the entire mass of pollutant placed in a monofill at the maximum predicted rate of loss for that pollutant is estimated. This approach is conservative because using higher estimates for pollutant flux leaving the monofill will yield a higher estimate of pollutant concentrations at the well.

For monofills, the rate of maximum total pollutant loss (in kg/yr) will occur in the year immediately following the last placement of sewage sludge, because the total mass of pollutant at the site reaches its peak at that time. As explained in Appendix C, this peak rate of loss could be maintained for a maximum length of time described by:

$$TP = \frac{LF}{\left[1 - e^{(-K_{la} LF)}\right]} \tag{4-96}$$

where:

TP = duration of "square wave" for approximating the loading of pollutant into the unsaturated soil zone (yr).

This result is combined with the estimate of the fraction of total pollutant loss through leaching for a conservative estimate of the average flux of pollutant leaching from the monofill:

$$FA_{lec,j} = \frac{10^{-6}\, C_j\, f_{la}\, SC}{TP} \tag{4-97}$$

where:

$FA_{lec,j}$ = annual average flux of pollutant j leaching from the monofill (kg pollutant/ha-yr),

C_j = concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge),

SC = estimated mass of sewage sludge contained in one hectare of completed monofill (kg/ha), and

10^{-6} = constant for converting units from (mg/ha-yr) to (kg/ha-yr).

Because sewage sludge is often combined with soil to increase solids content when placed in a monofill, the volume (and mass) of sewage sludge in the monofill is only a fraction of the total volume of the monofill. Therefore, the dry mass of sewage sludge contained in one hectare of filled monofill is calculated by multiplying the monofill's depth by the fraction of its volume containing sewage sludge and by the mass of solids in one cubic meter of sewage sludge:

$$SC = d_{mf} f_{sl} BD_{sludge} 10^4 \tag{4-98}$$

where:

$$BD_{sludge} = \frac{f_{sol} \rho_{sl} \rho_w}{f_{sol} \rho_w + (1 - f_{sol}) \rho_{sl}} \tag{4-99}$$

and:

BD_{sludge} = bulk density of sewage sludge (kg sewage sludge/m^3 sewage sludge),

f_{sl} = fraction of monofill's volume containing sewage sludge (dimensionless),

ρ_{sl} = particle density of sewage sludge (kg sewage sludge/m^3 sewage sludge),

ρ_w = density of water (kg water/m^3 water),

f_{sol} = fraction of solids in sewage sludge (kg solids/kg sewage sludge), and

10^4 = constant for converting units from (kg/m^2) to (kg/ha).

Next, dividing this estimated flux by the assumed net recharge and adjusting units yields the estimated average concentration of pollutant in leachate:

$$C_{lec,j} = \frac{0.1 \; FA_{lec,j}}{NR} \tag{4-100}$$

where:

$C_{lec,j}$ = average concentration of pollutant j in water leaching from the monofill site (mg pollutant/L water), and

0.1 = constant to convert from (kg/ha-m) to (mg/L).

The next step is to relate the leachate concentration to the expected concentration of pollutant in drinking water wells near the site. Two mathematical models are combined to calculate an expected ratio between these two concentrations. The Vadose Zone Flow and Transport finite element module (VADOFT) from the RUSTIC model (U.S. EPA, 1989d,g) is used to estimate flow and transport through the unsaturated zone, and the AT123D

analytical model (Yeh, 1981) is used to estimate pollutant transport through the saturated zone.

VADOFT allows consideration of multiple soil layers, each with homogeneous soil characteristics. Within the unsaturated zone, the attenuation of organic pollutants is predicted based on longitudinal dispersion, an estimated retardation coefficient derived from an equilibrium soil-water partition coefficient, and a first-order rate of pollutant degradation. The input requirements for the unsaturated zone module include various site-specific and geologic parameters and the leakage rate from the bottom of the monofill. It is assumed that the flux of pollutant mass into the top of the unsaturated zone beneath a monofill can be represented by results from the mass-balance calculations described above. Results from analysis of the unsaturated zone give the flow velocity and concentration profiles for each pollutant. These velocities and concentrations are evaluated at the water table, converted to a mass flux, and used as input to the AT123D saturated zone module.

The flow system in the vertical column is solved with VADOFT, which is based on an overlapping representation of the unsaturated and saturated zones. The water flux at the soil/liquid interface is specified for the bottom of the monofill, which defines the top of the unsaturated zone in the model. In addition, a constant pressure-head boundary condition is specified for the bottom of the unsaturated zone beneath the monofill. This pressure-head is chosen to be consistent with the expected pressure head at the bottom of the saturated zone, without consideration of the added flux seeping from sewage sludge in the monofill. Transport in the unsaturated zone is determined using the Darcy velocity and saturation profiles from the flow simulation. From these, the transport velocity profile can be determined.

Although limited to one-dimensional flow and transport, the use of a rigorous finite-element model in the unsaturated zone allows consideration of depth-variant physical and chemical processes that would influence the mass flux entering the saturated zone. Among the more important of these processes are advection (which is a function of the Darcy velocity, saturation, and porosity), mass dispersion, adsorption of the leachate onto the solids phase, and both chemical and biological degradation.

To represent the variably saturated soil column beneath the floor of the monofill, the model discretizes the column into a finite-element grid consisting of a series of one-dimensional elements connected at nodal points. Elements can be assigned different properties for the simulation of flow in a heterogenous system. The model generates the grid from user-defined zones; the user defines the homogeneous properties of each zone, the zone thickness, and the number of elements per zone, and the code automatically divides each zone into a series of elements of equal length. The governing equation is approximated using the Galerkin finite element method and then solved iteratively for the dependent variable (pressure-head), subject to the chosen initial and boundary conditions. Solution of the series of nonlinear simultaneous equations generated by the Galerkin scheme is accomplished by either Picard iteration, a Newton-Raphson algorithm or a modified Newton-Raphson algorithm. Once the finite-element calculation converges, the model yields estimated values for all the variables at each of the discrete nodal points. A detailed description of the solution scheme is found in U.S. EPA (1989g).

One-dimensional, advective-dispersive transport is estimated with VADOFT based on the estimated mass flux of pollutant into the top of the soil column, and a zero concentration boundary condition at the bottom of the saturated zone. As discussed earlier, sewage sludge is assumed to be deposited in the monofill for 20 years, followed by an inactive period in which pollutant is depleted from the monofill by leaching, volatilization, and degradation. To simulate potential contamination of groundwater, the loading of pollutant into the unsaturated zone beneath the monofill is "linearized" into a pulse of constant magnitude (TP) to represent the maximum annual loss of pollutant (in kg/ha-yr) occurring over the 300-year simulation period modeled. The duration of that pulse is calculated so that pollutant mass is conserved.

As in calculations for the unsaturated zone, degradation of organic pollutants is assumed to be first-order during transport through the aquifer. Speciation and complexation reactions are ignored for metals, leading to the possible over- or under-estimation of expected concentrations of metals in groundwater at the location of a receptor well. Detailed descriptions of the AT123D model are provided by U.S. EPA (1986d) and by Yeh (1981) and will not be repeated here. In general, the model provides an analytical solution to the basic advective-dispersive transport equation. One advantage of AT123D is its flexibility: the model allows the user up to 450 options and is capable of simulating a wide variety of configurations of source release and boundary conditions. For the current application, AT123D uses the source term and other input parameters to predict concentrations of pollutant (C_{wel}) within 300 years in a receptor well at the downgradient edge of the site's property boundary.

Method for Volatilization Pathway

Two steps provide an estimate of the concentration of a volatilized pollutant in air near the monofill:

1) Use the mass balance calculations summarized above to determine the mass of pollutant expected to volatilize from the monofill within a period equivalent to a human lifespan, and

2) Use a simplified version of the Industrial Source Complex Long Term Model (ISCLT) to model the transport and dispersion of pollutant in ambient air near the monofill.

Results from the mass balance calculations are used to estimate the fraction of total pollutant mass expected to volatilize from the monofill within an expected human lifetime (assumed to be 70 years), which spans both the active and inactive phases of the monofill's operation:

$$f_{vls} = f_{va} f_{ac} + f_{vi}(1 - f_{ac})\left[1 - e^{-K_{ti}(LS - LF)}\right] \tag{4-101}$$

where:

f_{vls}	=	fraction of pollutant mass that volatilizes over a human lifetime (dimensionless), and
LS	=	average human lifetime (yr).

Next, this fraction is multiplied by the total mass of pollutant placed in the monofill, and divided by the time of release to calculate an average flux:

$$FA_{vol,j} = \frac{10^{-6}\ C_j\ f_{vls}\ SC}{LS} \tag{4-102}$$

where:

$FA_{vol,j}$	=	annual average flux of pollutant j volatilizing from the monofill (kg pollutant/ha-yr),
10^{-6}	=	constant to convert units from (mg/ha-yr) to (kg/ha-yr), and
C_j	=	concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge).

The next step is to relate releases of volatilized pollutant to the expected concentrations in ambient air. The model used to simulate transport of pollutant from a monofill site is described by U.S. EPA (1986d) and is based on equations provided by Environmental Science and Engineering (1985). These equations are simplifications of equations used in ISCLT. The exposed individual is assumed to live at the downwind property boundary of the monofill site. A source-receptor ratio is calculated to relate the concentration of pollutant in ambient air at that individual's location (g pollutant/m^3 air) to the rate at which that pollutant is emitted from the monofill (g pollutant/m^2 monofill area-sec):

$$SRR = 2.032\ \frac{A_{mono} \cdot v}{\left(r' + x_y\right)\ u\ \sigma_z} \tag{4-103}$$

where:

SRR	=	source-receptor ratio (sec/m),
2.032	=	empirical constant,
A_{mono}	=	area of monofill (m^2),
v	=	vertical term for dispersion of pollutant in air (dimensionless),
r'	=	distance from the monofill's center to the downwind edge (m),
x_y	=	lateral virtual distance to the monofill (m),
u	=	average wind speed (m/sec), and
σ_z	=	standard deviation of the vertical distribution of pollutant concentration in air (m).

The vertical term (v) is a function of source height, the mixing layer height, and σ_z. Under stable conditions the mixing layer height is assumed to be infinite, and for a pollutant release height of zero, $v=1$. The lateral virtual distance is the distance from a virtual point source to the monofill, such that the angle θ subtended by the monofill width is 22.5°. This distance is calculated as:

$$x_y = \sqrt{\frac{A_{mono}}{\pi}} \cot \frac{\theta}{2} \tag{4-104}$$

The distance from the center of the land application site to the downwind edge is calculated assuming a square monofill:

$$r' = \frac{\sqrt{A_{mono}}}{2} \tag{4-105}$$

The standard deviation of the vertical distribution of concentration (σ_z) is defined by an atmospheric stability class and the distance from the center of the monofill to the downwind edge. Exhibit 4-18 provides values for two parameters, a and b, for a range of distances under stable atmospheric conditions. Based on values from this table, an appropriate value of σ_z is calculated as:

$$\sigma_z = a\, x^b \tag{4-106}$$

where:

$$x = 10^{-3} \cdot r' \tag{4-107}$$

and:

x	=	distance from the center of the monofill to the downwind edge (km), and
10^{-3}	=	constant to convert (m) to (km).

This result is combined with the estimated average flux of pollutant to predict the average concentration of pollutant in ambient air over this period:

$$C_{air,j} = FA_{vol,j} \cdot SRR \cdot 0.00317 \tag{4-108}$$

where:

$C_{air,j}$	=	average concentration of pollutant j in ambient air at the downwind edge of site (μg pollutant/m^3 air), and
0.00317	=	constant to convert (kg/ha-yr) to (μg/m^2-sec).

4.3.3 Methods for the Surface Impoundment Prototype

The methods for estimating exposure for surface impoundments are similar to those described in Section 4.3.2 for monofills. As with monofills, a mass balance of pollutant losses from the surface impoundment is calculated first.

Method for Mass Balance

Pollutants in sewage sludge are assumed to enter the surface impoundment through continuous inflow, and to be removed through four general processes:

1) degradation within the surface impoundment (e.g., photolysis, hydrolysis, or microbial decay);

2) transportation out of the surface impoundment by seepage through the floor of the impoundment;

3) outflow (possibly for return to the treatment works); and

4) volatilization from the liquid surface of the impoundment.

The model for describing these four processes in this analysis is adapted from a two-layer model suggested by Thomann and Mueller (1987) for modeling toxic substances in a lake. For the water column of a lake, those authors consider the inflow and outflow of pollutant, diffusive exchange between the solids layer and the water column, degradation, volatilization, the settling of particulate toxicant from the water column to the solid, and the re-suspension of particulate from the solids layer to the water column. For the solids layer, they consider diffusive exchange with the water column, decay processes, particulate settling from the overlying water column, re-suspension from the solids to the water column, and loss of toxicant from the solids due to net sedimentation or burial.

A similar, two-layer model is used for surface impoundments. The "liquid" layer begins at the surface and has the same average solids content as inflow to the surface impoundment; the "solid" layer beneath has a higher solids content. Although a gradient of solids concentrations is likely to form in an actual impoundment, each layer is idealized as homogeneous for both solids and pollutant concentrations.

Thomann and Mueller provide explicit equations for predicting settling velocities for particulates and rates of diffusive exchange between the two layers, but the present methodology derives simpler equations by assuming the solids layer will eventually reach the surface of the impoundment and outflow contains negligible concentrations of suspended solids. All loss processes are approximated as proportional to pollutant concentration; i.e., loss rates at any time are proportional to the current concentration of pollutant in the impoundment.

Two additional simplifying assumptions are made:

1) Concentrations of pollutant within each layer are assumed to be at steady-state and to be partitioned at equilibrium between adsorbed and dissolved phases.

2) Rates of pollutant transfer and loss when the impoundment is half-filled with solids are assumed to be typical of the surface impoundment both before and after it fills with sewage sludge.

If rates of loss to effluent, volatilization, seepage, and degradation are all proportional to pollutant concentration, the maximum total rate of loss will occur if equilibrium concentrations are attained. Moreover, after the continuous placement of sewage sludge in the surface impoundment is terminated, the rates at which pollutant is lost to seepage and volatilization should decline. By assuming that equilibrium conditions represent the entire (active and inactive) lifetime of the surface impoundment, this methodology probably overestimates rates of pollutant loss through seepage and volatilization, leading to conservative estimates of risks.

Liquid Layer. The concentration of pollutant in the inflow of the impoundment (C_i) and in the liquid layer (C_1) are assumed to remain constant throughout the surface impoundment's active lifetime. The partitioning of pollutant in the liquid layer is described as:

$$Q_i C_i = Q_o f_{d1} C_1 + K_{deg1} A d_1 C_1 + K_{vol1} f_{d1} A C_1 + Q_{sep} f_{d1} A C_1 + DVC_1 \qquad (4\text{-}109)$$

where:

Q_i	=	rate at which sewage sludge enters the impoundment (m^3 sewage sludge/sec),
C_i	=	concentration of pollutant in inflow to the impoundment (kg pollutant/m^3 sewage sludge),
Q_o	=	rate at which outflow leaves the impoundment, possibly for further treatment (m^3 sewage sludge/sec),
f_{d1}	=	fraction of total pollutant in liquid layer that is dissolved (dimensionless),
C_1	=	total concentration of pollutant (adsorbed and dissolved) in liquid layer of impoundment (kg pollutant/m^3 liquid layer),
K_{deg1}	=	anaerobic rate of pollutant degradation in liquid layer (sec^{-1}),
A	=	area of surface impoundment (m^2),
d_1	=	depth of liquid layer (m),
K_{vol1}	=	rate of pollutant volatilization from liquid layer (m/sec),
Q_{sep}	=	rate of seepage beneath the impoundment (m/sec), and
DV	=	rate of change in the volume of the liquid layer (m^3 liquid layer/sec).

Because the total depth of the impoundment (including both liquid and solids layers) is assumed constant, the depth of the liquid layer is reduced as more sewage sludge accumulates in the solids layer. If the rate at which the solids accumulates is constant over the active lifetime of the surface impoundment, the rate of accumulation can be determined by dividing the total depth of the impoundment by its expected active lifetime:

$$DV = \frac{d_{si}A}{TF} \tag{4-110}$$

where:

d_{si}	=	total depth of surface impoundment (m), and
TF	=	estimated active lifetime of surface impoundment (sec).

The active lifetime of the surface impoundment is calculated as:

$$TF = \frac{d_{si}AS_2}{Q_iS_1} \tag{4-111}$$

where:

S_1	=	concentration of solids in liquid layer (kg/m^3), and
S_2	=	concentration of solids in solids layer (kg/m^3).

For the first term on the right of Eq. 4-109, the volume of outflow from the surface impoundment (Q_o) is calculated to be consistent with assumptions about rates of inflow, seepage, and accumulation of the solids layer:

$$Q_o = Q_i\left(1 - \frac{S_1}{\rho_{sl}}\right) - Q_{sep}A - DV\left(1 - \frac{S_2}{\rho_{sl}}\right) \tag{4-112}$$

The concentration of solids in the liquid and solids layers is calculated from parameters describing the fraction of solids (by mass) in each layer:

$$S_1 = \frac{\rho_{sl}\rho_w P_1 10^3}{\rho_w P_1 10^3 + (1-P_1)\rho_{sl}} \qquad S_2 = \frac{\rho_{sl}\rho_w P_2 10^3}{\rho_w P_1 10^3 + (1-P_2)\rho_{sl}} \tag{4-113}$$

where:

P_1	=	fraction of solids (by mass) in liquid layer (kg solids/kg liquid layer),
P_2	=	fraction of solids (by mass) in solids layer (kg solids/kg solids layer),
ρ_w	=	density of water (kg water/L water),
ρ_{sl}	=	particle density of sewage sludge (kg sewage sludge/m^3 sewage sludge), and

10^3 = constant to convert (L) to (m^3).

In both the liquid and solids layers, a pollutant is partitioned between adsorbed and dissolved phases. The partitioning depends on both the pollutant-specific partition coefficient and the concentration of solids in the layer:

$$f_{dl} = \frac{1}{1 + K_d \, 10^{-3} \, S_1} \tag{4-114}$$

where:

f_{dl} = fraction of total pollutant in liquid layer that is dissolved (dimensionless),
K_d = soil-water partition coefficient (L water/kg soil), and
10^{-3} = constant to convert (L) to (m^3).

The second term on the right side of Eq. 4-109 describes degradation of the pollutant through photolysis, hydrolysis, microbial decay, and other processes. Values for K_{degl} are taken from studies of hydrolysis and microbial degradation, and are applied to pollutant in both dissolved and adsorbed phases.

The third term on the right side of Eq. 4-109 describes pollutant loss through volatilization, and is the only term directly linked with human exposure. The overall mass transfer coefficient for volatilization (K_{voll}) is calculated with a two-film resistance model (Thomann and Mueller, 1987) in which the overall resistance equals the sum of the liquid and gas phase resistances:

$$\frac{1}{K_{voll}} = \frac{1}{K_l} + \frac{RT \cdot 10^{-3}}{HK_g} \tag{4-115}$$

where:

K_{voll} = rate of pollutant volatilization from liquid layer (m/sec),
K_l = mass transfer coefficient for the liquid layer (m/sec),
K_g = mass transfer coefficient for the gas layer (m/sec),
R = gas constant (L-atm/mol-K),
T = average air temperature (Kelvin),
H = Henry's Law constant for pollutant (atm-m^3/mol), and
10^{-3} = constant to convert (L) to (m^3).

Numerous methods for calculating K_l and K_g for water surfaces have been proposed (see for example: Hwang, 1985; Mackay and Leinonen, 1975; Mackay and Yeun, 1983; Shen, 1982; Springer et al., 1984; U.S. EPA, 1987b; U.S. EPA, 1989e). This methodology follows an approach described in U.S. EPA (1987b, 1989e) for estimating volatilization from

surface impoundments. The selection of appropriate equations for calculating mass transfer coefficients depends on two characteristics of the site: (1) the ratio of the impoundment's effective diameter (or "fetch") to its depth and (2) the local average wind speed. Effective diameter (in meters) is defined as the diameter of a circle with area equal to that of the impoundment. Depth is defined as that of the liquid layer, which for the purpose of this calculation is assumed to average half of the impoundment's total depth. The ratio of fetch to depth is therefore calculated as:

$$de = 2\sqrt{A/\pi}$$
$$FR = \frac{de}{d_1} \qquad (4\text{-}116)$$

where:

2	=	factor to convert radius to diameter,
A	=	area of surface impoundment (m^2),
de	=	effective diameter (or fetch) of surface impoundment (m),
FR	=	ratio of fetch to depth (dimensionless), and
d_1	=	depth of liquid layer (m).

For surface impoundments where the average wind speed 10 m above the liquid surface is greater than 3.25 m/s and FR $\geq$ 51.2 (as in the scenario used for the surface impoundment prototype):

$$K_l = 2.611\times10^{-7}\, u_{10}^2 \left[\frac{D_{cw}}{D_{eth}}\right]^{2/3} \qquad (4\text{-}117)$$

where:

2.611×10^{-7}	=	empirical constant,
u_{10}	=	average wind speed 10 m above surface (m/sec),
D_{cw}	=	diffusivity of pollutant in water (cm^2/sec), and
D_{eth}	=	diffusivity of diethyl ether in water (8.5×10^{-6} cm^2/sec).

Calculation of the mass transfer coefficient for the gas phase is based on Hwang (1982). For all values of FD and u_{10}, the mass transfer coefficient for the gas layer is calculated from:

$$K_g = 1.8\times10^{-3} u_{10}^{0.78}\, Sc_G^{-0.67}\, de^{-0.11} \qquad (4\text{-}118)$$

where:

K_g	=	mass transfer coefficient for the gas layer (m/sec),
1.8×10^{-3}	=	empirical constant, and
Sc_G	=	the Schmidt number on the gas side (dimensionless), defined below.

$$Sc_G = \frac{\mu_a}{\rho_a D_{ca}} \tag{4-119}$$

where:

μ_a	=	viscosity of air (g/cm-sec),
ρ_a	=	density of air (g air/cm^3 air), and
D_{ca}	=	molecular diffusivity of pollutant in air (cm^2/sec).

Equations 4-117 and 4-118 are sufficient to estimate K_{voll}, the overall mass transfer coefficient for the dissolved fraction of the pollutant.

The fourth term on the right side of Eq. 4-109 describes losses of dissolved pollutant from the liquid layer as a result of the seepage through the solids layer and the floor of the impoundment. The rate of seepage (Q_{sep}) is based on measured values from sewage sludge lagoons. Only dissolved pollutant concentrations are included in this term; adsorbed pollutant concentrations are included in the fifth term of the equation, which describes loss of pollutant from the liquid layer as a result of the diminishing volume of that layer.

All terms on the right side of Eq. 4-109 are proportional to the concentration of pollutant in the liquid layer. A coefficient for the total rate at which pollutant mass is lost from the liquid layer (K_{tot1}, in m^3/sec) can be defined as:

$$K_{tot1} = Q_o f_{d1} + K_{deg1} d_1 A + K_{voll1} f_{d1} A + Q_{sep} f_{d1} A + DV \tag{4-120}$$

so that:

$$Q_i C_i = K_{tot1} C_1 \tag{4-121}$$

Because all estimated rates of pollutant loss are proportional to the concentration of pollutant in the liquid layer, total losses can be partitioned among competing loss processes according to fixed ratios. Of the total mass of pollutant lost from the liquid layer, the fraction lost to each process is:

$$f_{out1} = \frac{Q_o f_{d1}}{K_{tot1}} \qquad f_{deg1} = \frac{K_{deg1} d_1 A}{K_{tot1}} \qquad f_{voll} = \frac{K_{voll1} f_{d1} A}{K_{tot1}}$$
$$f_{sep1} = \frac{Q_{sep} f_{d1} A}{K_{tot1}} \qquad f_{del1} = \frac{DV}{K_{tot1}} \tag{4-122}$$

where:

f_{out1}	=	fraction of total pollutant lost from liquid layer that is lost in outflow from the impoundment (dimensionless),

f_{deg1} = fraction of total pollutant lost from liquid layer that is lost to degradation (dimensionless),

f_{vol1} = fraction of total pollutant lost from liquid layer that is lost to volatilization (dimensionless),

f_{sep1} = fraction of total pollutant lost from liquid layer that is lost to seepage (dimensionless), and

f_{del1} = fraction of total pollutant lost from the liquid layer as a result of the diminishing volume of the liquid layer (dimensionless).

Solids Layer. Pollutant mass accumulates in the solids layer as the depth of this layer increases and eventually reaches the surface of the impoundment. If the only source of pollutant mass for the solids layer is the loss estimated for the liquid layer, then:

$$Q_{sep}f_{d1}AC_1 + DVC_1 = K_{deg2}d_2AC_2 + Q_{sep}f_{d2}AC_2 + DVC_2 \qquad (4\text{-}123)$$

where:

f_{d2} = fraction of total pollutant in solids layer that is dissolved (dimensionless),

d_2 = depth of solids layer (m),

K_{deg2} = anaerobic rate of pollutant degradation in solids layer (sec^{-1}), and

C_2 = total concentration of pollutant in solids layer (kg pollutant/m^3 solids layer).

Similar to the liquid layer, the partitioning of a pollutant in the solids layer can be expressed as:

$$f_{d2} = \frac{1}{1 + K_d \cdot 10^{-3} \cdot S_2} \qquad (4\text{-}124)$$

where:

K_d = soil-water partition coefficient (L water/kg soil),

S_2 = concentration of solids in solids layer (kg/m^3), and

10^{-3} = constant to convert (L) to (m^3).

A coefficient for the total loss or storage of pollutant in the solids layer (K_{tot2}, in m^3/sec) can be defined as:

$$K_{tot2} = K_{deg2}d_2A + Q_{sep}f_{d2}A + DV \qquad (4\text{-}125)$$

As with the liquid layer, this coefficient can be partitioned into its individual components:

$$f_{deg2} = \frac{K_{deg2} d_2 A}{K_{tot2}} \qquad f_{sep2} = \frac{Q_{sep} f_{d2} A}{K_{tot2}} \qquad f_{del2} = \frac{DV}{K_{tot2}} \tag{4-126}$$

where:

f_{deg2} = fraction of pollutant reaching the solids layer that is lost to degradation (dimensionless),

f_{sep2} = fraction of pollutant reaching the solids layer that is lost to seepage (dimensionless), and

f_{del2} = fraction of pollutant reaching the solids layer that is stored in the accumulating depth of this layer (dimensionless).

If concentrations of pollutant in the liquid and solids layers can be approximated as steady-state for the duration of the impoundment's active lifetime, and if the partitioning of pollutant among competing loss processes halfway through the impoundment's active lifetime is assumed typical of its entire active phase, then the fraction of each year's loading of pollutant lost during each year of the surface impoundment's active phase can be calculated as:

$$f_{act} = f_{vol1} + f_{deg1} + f_{out1} + (f_{sep1} + f_{del1})(f_{deg2} + f_{sep2}) \tag{4-127}$$

where:

f_{act} = fraction of each year's loading of pollutant lost during each year of the surface impoundment's active phase (dimensionless).

Finally, if all pollutant is eventually lost from the impoundment and the partitioning of pollutant mass halfway through the surface impoundment's lifetime is generalized for the entire mass of pollutant, the fraction of pollutant mass lost through each pathway can be calculated as:

$$f_{sep} = \frac{(f_{sep1} + f_{del1})\, f_{sep2}}{f_{act}}$$

$$f_{vol} = \frac{f_{vol1}}{f_{act}}$$

$$f_{deg} = \frac{f_{deg1} + (f_{sep1} + f_{del1})\, f_{deg2}}{f_{act}} \qquad (4\text{-}128)$$

$$f_{out} = \frac{f_{out1}}{f_{act}}$$

where:

f_{sep}	=	fraction of total pollutant lost from the impoundment through seepage (dimensionless),
f_{vol}	=	fraction of total pollutant lost from the impoundment through volatilization (dimensionless),
f_{deg}	=	fraction of total pollutant lost from the impoundment through degradation (dimensionless), and
f_{out}	=	fraction of total pollutant lost from the impoundment through outflow (dimensionless).

These results are used to calculate concentrations of pollutant in groundwater and air near the surface impoundment.

Method for Groundwater Pathway

The methods for estimating concentrations of pollutants in groundwater near a surface impoundment are almost identical to those discussed above for monofills. First the concentration of pollutant in sewage sludge is used to estimate the expected flux of pollutant into the top of the unsaturated zone. To simplify the calculations, this pollutant flux is represented as a pulse of constant magnitude or "square wave," with its duration calculated so that the entire mass of pollutant will be depleted at the equilibrium rates calculated for the active impoundment:

$$TP = \frac{TF}{f_{act} \cdot 31{,}536{,}000} \qquad (4\text{-}129)$$

where:

TP	=	duration of "square wave" for approximating the loading of pollutant into the unsaturated soil zone (yr),
TF	=	estimated active lifetime of surface impoundment (sec), and
31,536,000	=	constant to convert (sec) to (yr).

This result is combined with another result from the mass balance calculations to derive a conservative estimate of the average flux of pollutant to the unsaturated zone beneath the site:

$$FA_{lec,j} = \frac{0.01\ f_{sep}\ S_2\ d_{si}\ C_j}{TP} \tag{4-130}$$

where:

$FA_{lec,j}$	=	annual average flux of pollutant j leaching through the floor of the surface impoundment (kg pollutant/ha-yr),
0.01	=	constant to convert (mg/m^2) to (kg/ha), and
C_j	=	concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge).

Next the average flux is used to estimate the average concentration of pollutant in seepage:

$$C_{sep,j} = \frac{0.1 \cdot FA_{lec,j}}{Q_{sep} \cdot 31{,}536{,}000} \tag{4-131}$$

where:

0.1	=	constant to convert (kg/ha-m) to (mg/L), and
$C_{sep,j}$	=	average concentration of pollutant j in water seeping through the bottom of the impoundment (mg pollutant/L water).

As discussed in Section 4.3.2 for the monofill prototype, two mathematical models are combined for this purpose. The VADOFT component of the RUSTIC model (U.S. EPA, 1989d,g) estimates flow and transport through the unsaturated zone, and the AT123D model (Yeh, 1981) estimates pollutant transport through the saturated zone.

Minor adjustments have been made to the linked models to represent a phenomenon unique to the surface impoundment prototype: seepage from a surface impoundment can cause local elevation of the water table if rates of seepage from the impoundment exceed natural rates of aquifer recharge in the surrounding area. Such elevation of the water table, or mounding, has two implications for the expected concentrations of sewage sludge pollutants at a receptor well. The first is that the reduced vertical distance between the impoundment and the local water table will result in decreased time of travel for water moving between the impoundment and the saturated zone. The second is that an increased hydraulic gradient will form in the aquifer between the surface impoundment and the downgradient receptor well. This change in the gradient will increase the expected rate of horizontal transport of the pollutant through the saturated zone.

To accommodate these two effects in the model calculations, an approach used in the RUSTIC model is modified. The first component (VADOFT) of the modified linked model performs calculations for a vertical column containing both unsaturated and saturated zones,

and predicts the extent to which the elevation of the water table will be increased by the flux of water from the impoundment. Once the vertical column problem has been solved for mass and water fluxes at the water table elevation, the second model component (AT123D) simulates the movement of pollutants through the saturated zone, with adjustments to represent increased elevation of the water table. Unlike RUSTIC, however, the present methodology does not allow for partial feedback between the unsaturated and saturated zone components of the model; the saturated zone is represented separately by an analytical transport model.

Saturated Zone. The AT123D model accepts as input the flux of pure pollutant mass entering the top of the saturated zone, and does not consider the extent of the pollutant's dilution by water from the source area, or the impact of that water on groundwater flow within the saturated zone. When the vertical movement of pollutant through the unsaturated zone is due only to infiltration throughout the area, the gradient within the aquifer is a function of the water entering the saturated zone, and neglect of the diluted state of the source term may be valid. For the case of a surface impoundment, however, neglect of the extent of the pollutant's original dilution could result in non-trivial overestimation of the source concentration, leading to an overestimation of pollutant concentrations at the receptor well. Furthermore, neglect of mounding effects could lead to incorrect assumptions about the velocity of groundwater flow near the site.

These concerns are addressed with three simple adjustments to the execution of the AT123D model. First, to correct for AT123D's potential overestimation of the original concentration of pollutant at the aquifer's boundary, the mass flux estimated from VADOFT results is adjusted by a dilution factor (D_f) as follows:

$$D_f = \frac{F_a}{A Q_{sep} + F_a} \tag{4-132}$$

where:

F_a = the volume of fluid passing through a vertical cross-section of the aquifer oriented perpendicular to the direction of flow, and having a width equal to the source width and a depth equal to the saturated thickness of the aquifer (m^3/sec).

The excess water released by seepage from a surface impoundment also can result in a superimposed radial velocity field on the background or regional velocity field of groundwater flow. In other words, the horizontal velocity of water within the aquifer can be decreased upgradient of the surface impoundment, and increased downgradient of the surface impoundment. This change in the velocity field might result in reduced time of travel for pollutants moving to receptor wells downgradient of the impoundment, which could in turn lead to reductions in pollutant degradation prior to human exposure. Accurate accounting of the influence of mixing and degradation would require a fully three-dimensional flow and transport model; this methodology uses a simpler approach to estimate a conservative limit to pollutant decay within the system. The limit is estimated by increasing the estimated velocity of groundwater flow to account for the maximum downgradient increase in velocity

due to the source. The velocity increase can be approximated by idealizing the surface impoundment as a circular source, so that the rate at which seepage passes outward through a cylinder beneath the perimeter of the impoundment is:

$$v_i = \frac{Q_{sep} de}{4 d_a} \tag{4-133}$$

where:

v_i = superimposed radial velocity from water seeping from impoundment (m/sec), and

d_a = depth of aquifer (m).

In addition to increasing the expected velocity of pollutant transport through the aquifer, this superimposed velocity also would have the effect of increasing AT123D's estimate of pollutant dilution within the aquifer. This additional dilution effect must be subtracted back out of the model calculations, because the true dilution is explicitly included in the factor introduced by Eq. 4-132. The model performs this calculation automatically, based on the following equation for the anti-dilution factor:

$$D_{af} = \frac{v_v + v_h}{v_h} \tag{4-134}$$

where:

D_{af} = anti-dilution factor (dimensionless),

v_v = vertical velocity due to the source (m/sec), and

v_h = regional velocity of horizontal groundwater flow (m/sec).

It should be noted that the above methodology is conservative, because it overestimates the velocity beneath the source and does not allow for decreases in the superimposed velocity beyond the source. As a result, the methodology is more conservative than a three-dimensional model. In comparison with a two-dimensional cross-sectional flow and transport model, the model is more conservative beneath the source, but less conservative beyond the source.

By combining the VADOFT model with AT123D, and by adjusting calculations in AT123D to accommodate the dilution and superimposed velocity described above, concentrations of a pollutant in groundwater at a receptor well can be predicted as a function of the liquid concentration of pollutant near the floor of the impoundment, the rate of seepage from the surface impoundment, and hydrogeological characteristics of the area. It should be noted that all of the calculations described above are linear with respect to pollutant concentrations in liquid seeping from the impoundment.

Method for Volatilization Pathway

Estimates of exposure for the vapor pathway are based on the highest average concentrations of pollutant to be encountered over an expected human lifetime. At the maximum rate at which pollutant is lost during the surface impoundment's active operation, the fraction that would be lost to all processes over a period equivalent to the life expectancy is:

$$f_{vls} = f_{act} \cdot f_{voll} \tag{4-135}$$

where:

f_{vls}	=	fraction of pollutant mass that volatilizes over a human lifetime (dimensionless),
f_{act}	=	fraction of each year's loading of pollutant lost during each year of the surface impoundment's active phase (dimensionless), and
f_{voll}	=	fraction of total pollutant lost from liquid layer that is lost to volatilization (dimensionless).

This fraction can be converted to an average flux of pollutant volatilizing from the site as:

$$FA_{vol,j} = \frac{C_j \, f_{vls} \; 0.01 \; S_2 \; d_{si}}{LS} \tag{4-136}$$

where:

$FA_{vol,j}$	=	annual average flux of pollutant j volatilizing from the surface impoundment (kg/ha-yr),
C_j	=	concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge),
f_{vls}	=	fraction of pollutant mass that volatilizes over a human lifetime (dimensionless),
0.01	=	constant to convert (mg/m^2) to (kg/ha),
S_2	=	concentration of solids in solids layer (kg/m^3),
d_{si}	=	total depth of impoundment (m), and
LS	=	average human lifetime (yr).

The next step is to relate releases of volatilized pollutant from the site to the expected concentration in ambient air. As before, the simplified version of ISCLT described above to calculate a source-receptor ratio (SRR) is used. Multiplying the SRR by the average volatilized flux and adjusting units yields a conservative estimate of the expected average concentration of pollutant in ambient air near the site:

$$C_{air,j} = \frac{FA_{vol,j} \cdot SRR}{315.36} \tag{4-137}$$

where:

$C_{air,j}$	=	average concentration of pollutant j in ambient air at the downwind edge of the site (μg pollutant/m^3 air),
SRR	=	source-receptor ratio (sec/m), and
315.36	=	constant to convert (kg/ha-yr) to (μg/m^2-sec).

4.3.4 Estimating Human Exposure

To estimate human exposure, the methods discussed in Sections 4.3.2 and 4.3.3 are used to estimate the concentrations of each pollutant in air and groundwater near each type of surface disposal site prototype. Estimated concentrations in environmental media are converted to estimates of human exposure based on assumptions about the rate at which the average individual inhales air and ingests drinking water. For air, human exposure is calculated as:

$$EXP_j = \frac{10^{-3} \cdot C_{air,j} \cdot IA}{BW} \tag{4-138}$$

where:

EXP_j	=	exposure to pollutant j in sewage sludge (mg pollutant/kg body weight-day),
10^{-3}	=	constant to convert units from (μg) to (mg),
$C_{air,j}$	=	average concentration of pollutant j in ambient air at the downwind edge of the site (μg pollutant/m^3 air),
IA	=	inhalation rate (m^3 air/day), and
BW	=	body weight (kg).

Potential exposure to pollutants through ingestion of groundwater is calculated as:

$$EXP_j = \frac{C_{wel,j} \cdot IW}{BW} \tag{4-139}$$

where:

$C_{wel,j}$	=	concentration of pollutant j in well water (mg pollutant/L water), and
IW	=	volume of water ingested daily (L water/day).

4.3.5 Data Inputs

Exhibits 4-21 and 4-22 present the non-pollutant-specific data for the monofill prototype and the surface impoundment prototype.

EXHIBIT 4-21
Site and Sewage Sludge Parameters for Monofill Prototype

Parameter	Value	Reference
Area of Monofill (m^2)	10,000	U.S. EPA, 1992c
Depth of Monofill (m)	3.46	U.S. EPA, 1992c
Distance to Well (m)	150	U.S. EPA, 1992c
Thickness of Daily Cover (m)	0.3	U.S. EPA, 1992c
Thickness of Permanent Cover (m)	1	U.S. EPA, 1992c
Time Each Unit Uncovered (hr)	12	U.S. EPA, 1992c
Time Average Unit Contains Sewage Sludge (hr)	87,660	U.S. EPA, 1992c
Sewage Sludge as Fraction of Total Volume (m^3/m^3)	0.63	U.S. EPA, 1992c
Active Monofill Life (yr)	20	U.S. EPA, 1992c
Average Wind Speed (m/sec)	4.5	U.S. EPA, 1992c
Average Air Temperature (Kelvin)	288	U.S. EPA, 1992c
Vertical Term for Pollutant Dispersion in Air (dimensionless)	1	U.S. EPA, 1992c
Net Recharge (m/yr)	0.5	U.S. EPA, 1992c
Solids Content of Sewage Sludge (kg/kg)	0.20	U.S. EPA, 1992c

EXHIBIT 4-22
Site and Sewage Sludge Parameters for Surface Impoundment Prototype

Parameter	Value	Reference
Area of Surface Impoundment (m^2)	20,236	U.S. EPA, 1992c
Total Depth of Surface Impoundment (m)	4	U.S. EPA, 1992c
Distance to Well (m)	150	U.S. EPA, 1992c
Rate of Inflow (m^3/sec)	0.0022	U.S. EPA, 1992c
Average Wind Speed (m/sec)	4.5	U.S. EPA, 1992c
Average Air Temperature (Kelvin)	288	U.S. EPA, 1992c
Solids Content of Inflow (kg/kg)	0.03	U.S. EPA, 1992c
Solids Content of Liquid Layer (kg/kg)	0.03	U.S. EPA, 1992c
Solids Content of Solids Layer (kg/kg)	0.175	U.S. EPA, 1992c
Particle Density of Sewage Sludge (kg/m^3)	1200	U.S. EPA, 1992c
Net Seepage Beneath Impoundment (m/yr)	2.5	U.S. EPA, 1992c
Depth of Solids Layer (m)	2	Assumed half-filled
Depth of Liquid Layer (m)	2	Assumed half-filled
Vertical Term for Pollutant Dispersion in Air (dimensionless)	1	U.S. EPA, 1992c
Density of Air at 15°C (g/cm^3)	1.226×10^{-3}	Weast, 1990
Viscosity of Air at 15°C (g/cm-sec)	1.79×10^{-4}	Interpolated from Henry and Heinke, 1989

There are several pollutant-specific fate and transport parameters required to maintain the mass balance of a pollutant among the three loss processes and to estimate the rates at which those three loss processes occur. In Exhibit 4-23the fate and transport parameters for both the monofill and surface impoundment prototypes are presented. For degradation rates, estimates of both aerobic and anaerobic degradation rates are presented. As in Round One, for the degradation rate of a given pollutant in the unsaturated zone, ten percent of the aerobic biodegradation rate is used if an hydrolysis rate was unavailable. For the degradation rate for a given pollutant in the saturated zone, the arithmetic average of the unsaturated zone degradation rate and the anaerobic degradation rate for that pollutant is used.

To obtain estimates of inorganic K_d values for six Round Two pollutants, studies of adsorption described in Gerritse et al. (1982) were used. Gerritse et al. present a range of K_d values for various inorganics in two soil types: sand and sandy loam. In the sandy soil, there was 0.035 g/g organic matter, 0 g/g clay, 0.22 meq/g cation exchange capacity (CEC), and the porewater had a pH of 5. In the sandy loam soil, there was 0.025 g/g organic matter, 0.2 g/g clay, 0.16 meq/g CEC, and the porewater had a pH of 8. For this analysis,

EXHIBIT 4-23
Environmental Fate and Transport Parameters

Pollutant	log K_{ow}	K_d (L/kg)	Henry's Law Constant (atm-m^3/mol)	Aerobic[1] K_{deg} (yr^{-1})	Anaerobic[1] K_{deg} (yr^{-1})	Diffusivity in Air $(D_{ca})^2$ (cm^2/sec)	Diffusivity in Water $(D_{cw})^3$ (cm^2/sec)
Acetic acid (2,4-dichlorophenoxy)	2.81[4]	4.4[5]	1.4×10^{-10} [6]	36[7]	2.5[42]	5.5×10^{-2}	4.9×10^{-6}
Aluminum	NA	4[8]	NA	NA	NA	NA	NA
Antimony	NA	9[10]	NA	NA	NA	NA	NA
Asbestos	NA	NA	NA	NA	NA	NA	NA
Barium	NA	31[10]	NA	NA	NA	NA	NA
Beryllium	NA	170[10]	NA	NA	NA	NA	NA
Bis(2-ethylhexyl) phthalate	5.11[12]	540[5]	8.8×10^{-6} [13]	11[14]	0[43]	3.4×10^{-2}	2.9×10^{-6}
Boron	NA		NA	NA	NA	NA	NA
Butanone, 2-	0.29[16]	0.022[5]	5.8×10^{-5} [17]	25[18]	9.0[14]	9.2×10^{-2}	7.8×10^{-6}
Carbon disulfide	2.00[19]	0.81[5]	1.4×10^{-2} [20]	0.006[21]	0.006[21]	1.1×10^{-1}	9.7×10^{-6}
Cresol, p-	1.94[4]	0.71[5]	7.9×10^{-7} [22]	380[14]	9.0[14,44]	7.4×10^{-2}	6.6×10^{-6}
Cyanide (soluble salts and complexes)	-1.04[23]	0.0014[5]	1.2×10^{-4} [24]	13[24]	13[24]	1.9×10^{-1}	1.8×10^{-5}
Dioxins and Dibenzofurans	6.64[25]	13,000[5]	6.8×10^{-5} [25]	0[25]	0[25]	4.4×10^{-2}	4.1×10^{-6}
Endosulfan-II	3.52[26]	19[5]	1.9×10^{-5} [27]	6.7[28]	6.7[28]	4.3×10^{-2}	3.6×10^{-6}

EXHIBIT 4-23
Environmental Fate and Transport Parameters (cont'd.)

Pollutant	log K_{ow}	K_d (L/kg)	Henry's Law Constant (atm-m^3/mol)	Aerobic[1] K_{deg} (yr^{-1})	Anaerobic[1] K_{deg} (yr^{-1})	Diffusivity in Air $(D_{ca})^2$ (cm^2/sec)	Diffusivity in Water $(D_{cw})^3$ (cm^2/sec)
Fluoride	NA	36[8]	NA	NA	NA	NA	NA
Manganese	NA	5[10]	NA	NA	NA	NA	NA
Methylene chloride	1.3[29]	0.19[5]	2.0×10^{-3} [27]	2.6[30]	12[30]	1.0×10^{-1}	9.2×10^{-6}
Nitrate	NA	NA	NA	NA	NA	NA	NA
Nitrite	NA	NA	NA	NA	NA	NA	NA
Pentachloronitrobenzene	4.64[4]	200[5]	9.9×10^{-5} [31]	1.1[32]	0[45]	5.2×10^{-2}	4.9×10^{-6}
Phenol	1.46[33]	0.26[5]	4.0×10^{-7} [22]	72[34]	18[46]	8.2×10^{-2}	7.4×10^{-6}
Polychlorinated biphenyls (coplanar)	6.7[35]	15,000[5]	2.0×10^{-4} [36]	0.063[37]	1.2[47]	4.5×10^{-2}	4.1×10^{-6}
Propanone, 2-	-0.24[4]	0.0074[5]	3.7×10^{-5} [38]	36[14]	250[48]	1.1×10^{-1}	9.1×10^{-6}
Propionic acid, 2-(2,4,5-trichlorophenoxy)	3.41[4]	15[5]	1.3×10^{-8} [39]	17[40]	0[45]	4.9×10^{-2}	4.4×10^{-6}
Silver	NA	539[10]	NA	NA	NA	NA	NA
Thallium	NA		NA	NA	NA	NA	NA
Tin	NA		NA	NA	NA	NA	NA
Titanium	NA		NA	NA	NA	NA	NA

EXHIBIT 4-23
Environmental Fate and Transport Parameters (cont'd.)

Pollutant	log K_{ow}	K_d (L/kg)	Henry's Law Constant (atm-m^3/mol)	Aerobic[1] K_{deg} (yr^{-1})	Anaerobic[1] K_{deg} (yr^{-1})	Diffusivity in Air $(D_{ca})^2$ (cm^2/sec)	Diffusivity in Water $(D_{cw})^3$ (cm^2/sec)
Toluene	2.79[27]	4.2[5]	6.7 x 10^{-3} [27]	120[41]	1.9[49]	8.0 x 10^{-2}	6.9 x 10^{-6}
Trichlorophenoxy acetic acid, 2,4,5-	3.13[4]	8.6[5]	9.4 x 10^{-11} [11]	13[9]	0.75[15]	5.4 x 10^{-2}	4.6 x 10^{-6}
Vanadium	NA	68[10]	NA	NA	NA	NA	NA

Notes: NA means Not Applicable
[1] Abiotic hydrolysis rates preferred over biodegradation rates.
[2] Estimated from Lyman et al. (1990), using the Wilke and Lee method.
[3] Estimated from Lyman et al. (1990), using the Hayduk and Laudie method.
[4] Hansch and Leo (1985a).
[5] Estimated using Eq. 4-57 and 4-58.
[6] Jury et al. (1987).
[7] Ou (1984).
[8] Based on Langmuir isotherm data in Bodek et al. (1988).
[9] Smith (1978, 1979); McCall et al. (1981); Alton and Stritzke (1973).
[10] Median value from K_d range in sand measured for that inorganic in Gerritse et al. (1982).
[11] Based on water solubility (Klopffer et al., 1982) and vapor pressure (Que Hee et al., 1981).
[12] CHEMEST procedure of GEMS, U.S. EPA (1989c).
[13] Howard et al. (1985); Montgomery and Welkom (1990).
[14] Howard et al. (1991).
[15] Sattar and Paasivirta, 1980.
[16] Hansch and Leo (1981).
[17] Rathburn and Tai (1987).
[18] Dojlido (1979); Price et al. (1974); Bridie et al. (1979); Dose et al. (1975); Heukelekian and Rand (1955).
[19] Verschueren (1983).
[20] Elliot (1989).
[21] Peyton et al. (1976).

[22] Hine and Mookerjee (1975).
[23] Estimated from Lyman et al. (1990) for hydrogen cyanide.
[24] Bodek et al. (1988).
[25] U.S. EPA (1994b).
[26] OHM/TADS (1989).
[27] Mabey et al. (1982).
[28] Greve and Wit (1971).
[29] Hansch and Leo (1979).
[30] Davis and Madsen (1991); anaerobic rate after a 70 day lag.
[31] Calculated from water solubility (Eckert, 1962) and vapor pressure (Gile and Gillett, 1979).
[32] Wang and Broadbent (1972).
[33] Hansch and Leo (1985b).
[34] Baker and Mayfield (1980).
[35] Based on Aroclor 1254 from O'Connor (1992) and Anderson and Parker (1990).
[36] Murphy et al. (1987).
[37] Fries (1982).
[38] Snider and Dawson (1985).
[39] Howard, 1991.
[40] Smith (1978); Alton and Stritzke (1973).
[41] Wilson et al. (1983).
[42] Liu et al., 1981.
[43] Horowitz et al., 1982; Shelton et al., 1984; Johnson and Lulves, 1975; Schwartz et al., 1979.
[44] Boyd et al., 1983.
[45] Default value due to lack of readily available data.
[46] Healy and young, 1978; Boyd et al., 1983.
[47] Rhee et al., 1989.
[48] Estimated from Chou et al., 1979.
[49] Horowitz et al., 1982 (average).

the K_d values from sand, which were lower than those in sandy loam, were used. The median K_d value from the range available for each of the six Round Two inorganics tested was used.

For aluminum and fluoride, available data on Langmuir isotherm parameters were used to estimate K_d values (Bodek et al., 1988). For aluminum, data were for silica, at a pH of 5. For fluoride, data corresponded to clay loam, containing 10.4 percent clay, 0.94 percent organic carbon, and 825 μg/g aluminum, with a pH of 5.9. For boron, thallium, tin, and titanium, K_d values were not available.

4.3.6 Modeling of Surface Impoundments in the Comprehensive Hazard Identification

For this risk assessment, only surface impoundments were modeled for the surface disposal option. Exposure to pollutants in surface impoundments is greater than exposure to pollutants in monofills for both the air and groundwater exposure pathways. Pollutants volatilize more readily from surface impoundments than from monofills because there is no soil cover. Pollutants also leach more readily from surface impoundments because the recharge rate to groundwater is higher due to the liquid sewage sludge being placed in the impoundment. Therefore, if a pollutant in a surface impoundment did not result in high risk levels, then its presence in a monofill also would not be of concern.

4.3.7 Example Exposure Calculations for Surface Disposal

The following example calculates exposure of humans to methylene chloride present in sewage sludge placed in a surface impoundment. Exposure occurs through ingestion of groundwater that has received leachate from a surface impoundment and through inhalation of methylene chloride that has volatilized from a surface impoundment.

Groundwater Exposure Pathway for Surface Impoundment

To estimate human exposure through the ground water pathway, computations must be made for both the liquid layer of the surface impoundment and the solids layer.

Calculations for Liquid Layer of Surface Impoundment. Several steps are necessary to estimate the rate at which methylene chloride is lost from the liquid layer of a surface impoundment. First, the mass transfer coefficients for the liquid layer of the surface impoundment and the air layer immediately above it are calculated. Equation 4-117 is used to calculate the mass transfer coefficient for the liquid layer:

$$K_l = 2.611x10^{-7} \cdot (4.5\,m/\text{sec})^2 \cdot \left[\frac{9.2x10^{-6}\,cm^2/\text{sec}}{8.5x10^{-6}\,cm^2/\text{sec}}\right]^{2/3}$$

$$= 5.6x10^{-6}\,m/\text{sec}$$

where:

$2.611x10^{-7}$	=	empirical constant,
4.5	=	u_{10} (average wind speed 10 m above surface) from Exhibit 4-22,
$9.2x10^{-6}$	=	D_{cw} (diffusivity of methylene chloride in water) from Exhibit 4-23, and
$8.5x10^{-6}$	=	D_{eth} (diffusivity of diethyl ether in water) from text.

To calculate the mass transfer coefficient for the gas layer immediately above the surface impoundment, the Schmidt number and the effective diameter of the surface impoundment must be calculated. The Schmidt number is calculated using Eq. 4-119:

$$Sc_G = \frac{1.79x10^{-4}\,g/cm\text{-sec}}{1.23x10^{-3}\,g/cm^3 \cdot 1.0x10^{-1}\,cm^2/\text{sec}}$$

$$= 1.5$$

where:

$1.79x10^{-4}$	=	μ_a (viscosity of air) from Exhibit 4-22,
$1.23x10^{-3}$	=	ρ_a (density of air) from Exhibit 4-22, and
$1.0x10^{-1}$	=	D_{ca} (diffusivity of methylene chloride in air) from Exhibit 4-23.

Equation 4-116 is used to calculate the effective diameter of the surface impoundment:

$$de = 2\sqrt{\frac{20{,}236\,m^2}{\pi}} = 160\,m$$

where:

2	=	factor to convert radius to diameter, and
20,236	=	A (area of surface impoundment) from Exhibit 4-22.

The mass transfer coefficient for the gas layer is then calculated using Eq. 4-118:

$$K_g = 1.8x10^{-3} \cdot 4.5^{0.78} \cdot 1.5^{-0.67} \cdot 160^{-0.11}$$

$$= 2.6x10^{-3} \frac{m}{\text{sec}}$$

where:

$1.8x10^{-3}$	=	empirical constant,
4.5	=	u_{10} (average wind speed 10 m above surface) from Exhibit 4-22,
1.5	=	Sc_G (Schmidt number on the gas side) calculated above, and
160	=	de (effective diameter of surface impoundment) calculated above.

Both K_g and K_l are then used in Equation 4-115 to calculate the overall mass transfer coefficient for volatilization:

$$\frac{1}{K_{vol1}} = \frac{1}{5.6x10^{-6} m/\text{sec}} + \frac{0.082(L-atm)/(mol-K) \cdot 288K \cdot 10^{-3} m^3/L}{2.0x10^{-3}(m^3 \cdot atm)/(mol) \cdot 2.6x10^{-3} m/\text{sec}}$$

$$= 1.8x10^5 \text{sec}/m$$

where:

$5.6x10^{-6}$	=	K_l (mass transfer coefficient for the liquid layer) calculated above,
0.082	=	R (gas constant) from Exhibit 4-16,
288	=	T (average air temperature) from Exhibit 4-22,
$2.0x10^{-3}$	=	H (Henry's Law constant for methylene chloride) from Exhibit 4-23, and
$2.6x10^{-3}$	=	K_g (mass transfer coefficient for the gas layer) calculated above.

Taking the inverse of $1/K_{vol1}$:

$$K_{vol1} = 5.4x10^{-6} m/\text{sec}$$

Second, the active lifetime of the surface impoundment and the rate of change in the volume of the liquid layer are calculated. The concentrations of solids in the liquid and solids layers must first be calculated by using Eq. 4-113:

$$S_1 = \frac{1200 kg/m^3 \cdot 1 kg/L \cdot 0.03 \cdot 1000 L/m^3}{1 kg/L \cdot 0.03 \cdot 1000 L/m^3 + (1-0.03) \cdot 1200 kg/m^3} = 30 kg/m^3$$

and:

$$S_2 = \frac{1200 kg/m^3 \cdot 1 kg/L \cdot 0.175 \cdot 1000 L/m^3}{1 kg/L \cdot 0.03 \cdot 1000 L/m^3 + (1-0.175) \cdot 1200 kg/m^3} = 206 kg/m^3$$

where:

1200	=	ρ_{sl} (particle density of sewage sludge) from Exhibit 4-22,
1	=	ρ_w (density of water) from Exhibit 4-16,
0.03	=	P_1 (fraction of solids (by mass) in liquid layer) from Exhibit 4-22,
0.175	=	P_2 (fraction of solids (by mass) in solids layer) from Exhibit 4-22, and
1000	=	constant to convert (L) to (m^3).

The active lifetime of the surface impoundment is then calculated using Eq. 4-111:

$$TF = \frac{4m \cdot 20{,}236 m^2 \cdot 206 kg/m^3}{0.0022 m^3/\text{sec} \cdot 30 kg/m^3}$$

$$= 2.5x10^8 \text{sec}$$

where:

4	=	d_{si} (total depth of surface impoundment) from Exhibit 4-22,
20,236	=	A (area of surface impoundment) from Exhibit 4-22,
206	=	S_2 (concentration of solids in solids layer) calculated above,
0.0022	=	Q_i (rate at which sewage sludge enters the surface impoundment) from Exhibit 4-22, and
30	=	S_1 (concentration of solids in liquid layer) calculated above.

The rate of change in the volume of the liquid layer is calculated using Eq. 4-110:

$$DV = \frac{4m \cdot 20{,}236m^2}{2.5x10^8 \sec}$$

$$= 3.2x10^{-4} m^3/\sec$$

where:

4	=	d_{si} (total depth of surface impoundment) from Exhibit 4-22,
20,236	=	A (area of surface impoundment) from Exhibit 4-22, and
$2.5x10^8$	=	TF (active lifetime of the surface impoundment) calculated above.

Third, the volume of outflow from the surface impoundment is calculated using Eq. 4-112:

$$Q_o = 0.0022 \frac{m^3}{\sec} \cdot \left(1 - \frac{30 \frac{kg}{m^3}}{1200 \frac{kg}{m^3}}\right) - 2.5 \frac{m}{yr} \cdot 20{,}236m^2 \cdot$$

$$\frac{1yr}{31{,}536{,}000 \sec} - 3.2x10^{-4} \frac{m^3}{\sec} \cdot \left(1 - \frac{206 \frac{kg}{m^3}}{1200 \frac{kg}{m^3}}\right)$$

$$= 2.7x10^{-4} \frac{m^3}{\sec}$$

where:

0.0022	=	Q_i (rate at which sewage sludge enters the surface impoundment) from Exhibit 4-22,
30	=	S_1 (concentration of solids in liquid layer) calculated above,
1200	=	ρ_{sl} (particle density of sewage sludge) from Exhibit 4-22,
2.5	=	Q_{sep} (rate of seepage beneath the impoundment) from Exhibit 4-22,
$3.2x10^{-4}$	=	DV (rate of change in the volume of the liquid layer) calculated above, and
206	=	S_2 (concentration of solids in solids layer) calculated above.

Fourth, Eq. 4-114 is used to calculate the fraction of methylene chloride dissolved in the liquid layer:

$$f_{dl} = \frac{1}{1 + 0.19\frac{L}{kg} \cdot 10^{-3}\frac{m^3}{L} \cdot 30\frac{kg}{m^3}} = 0.99$$

where:

0.19	=	K_d (soil-water partition coefficient for methylene chloride) from Exhibit 4-23,
10^{-3}	=	constant to convert (L) to (m^3), and
30	=	S_l (concentration of solids in liquid layer) calculated above.

Fifth, the total rate at which methylene chloride is lost from the liquid layer of a surface impoundment can be calculated using Eq. 4-120:

$$K_{totl} = (2.7x10^{-4} m^3/\text{sec} \cdot 0.99) + (12/yr \cdot \frac{1\,yr}{31{,}536{,}000\,\text{sec}} \cdot 2m \cdot 20{,}236\,m^2)$$

$$+ (5.4x10^{-6} m/\text{sec} \cdot 0.99 \cdot 20{,}236\,m^2) +$$

$$(2.5\,m/yr \cdot \frac{1\,yr}{31{,}536{,}000\,\text{sec}} \cdot 0.99 \cdot 20{,}236\,m^2) + (3.2x10^{-4} m^3/\text{sec})$$

$$= 0.13\,m^3/\text{sec}$$

where:

$2.7x10^{-4}$	=	Q_o (rate at which outflow leaves the impoundment) calculated above,
0.99	=	f_{dl} (fraction of total methylene chloride in liquid layer that is dissolved) calculated above,
12	=	K_{degl} (anaerobic degradation rate for methylene chloride) from Exhibit 4-23,
2	=	d_l (depth of liquid layer) from Exhibit 4-22,
20,236	=	A (area of surface impoundment) from Exhibit 4-22,
$5.4x10^{-6}$	=	K_{vol1} (rate of volatilization of methylene chloride from liquid layer) calculated above,
2.5	=	Q_{sep} (rate of seepage beneath the impoundment) from Exhibit 4-22, and
$3.2x10^{-4}$	=	DV (rate of change in the volume of the liquid layer) calculated above.

The last calculations pertaining to the liquid layer estimate the fractions of total methylene chloride lost from the liquid layer to seepage and as a result of the diminishing volume of the liquid layer as the surface impoundment is filled. Eq. 4-122 is used to calculate the fraction of total methylene chloride lost from the liquid layer to seepage:

$$f_{sepl} = \frac{2.5\,m/yr \cdot \frac{1\,yr}{31{,}536{,}000\,sec} \cdot 0.99 \cdot 20{,}236\,m^2}{0.13\,m^3/sec}$$

$$= 1.3x10^{-2}$$

where:

2.5	=	Q_{sep} (rate of seepage beneath the impoundment) from Exhibit 4-22,
0.99	=	f_{dl} (fraction of total methylene chloride in liquid layer that is dissolved) calculated above,
20,236	=	A (area of surface impoundment) from Exhibit 4-22, and
0.13	=	K_{totl} (coefficient for the total rate at which methylene chloride is lost from the liquid layer) calculated above.

Eq. 4-122 is also used to calculate the fraction of total methylene chloride lost from the liquid layer due to the diminishing volume of the liquid layer:

$$f_{dell} = \frac{3.2x10^{-4}\,m^3/sec}{0.13\,m^3/sec} = 2.5x10^{-3}$$

where:

$3.2x10^{-4}$	=	DV (rate of change in the volume of the liquid layer) calculated above, and
0.13	=	K_{totl} (coefficient for the total rate at which methylene chloride is lost from the liquid layer) calculated above.

Calculations for Solids Layer of Surface Impoundment. As for the liquid layer of the surface impoundment, several computational steps are required for the solids layer. First, the fraction of total methylene chloride that is dissolved in the solids layer is calculated using Eq. 4-124:

$$f_{d2} = \frac{1}{1 + 0.19\frac{L}{kg} \cdot 10^{-3}\frac{m^3}{L} \cdot 206\frac{kg}{m^3}} = 0.96$$

where:

0.19	=	K_d (soil-water partition coefficient for methylene chloride) from Exhibit 4-23,
10^{-3}	=	constant to convert (L) to (m^3), and
206	=	S_2 (concentration of solids in solids layer) calculated above.

Second, the coefficient for the total loss or storage of methylene chloride in the solids layer is calculated using Eq. 4-125:

$$K_{tot2} = (12/yr \cdot \frac{1\,yr}{31{,}536{,}000\,\text{sec}} \cdot 2m \cdot 20{,}236\,m^2) +$$

$$(2.5\,m/yr \cdot \frac{1\,yr}{31{,}536{,}000\,\text{sec}} \cdot 0.96 \cdot 20{,}236\,m^2) + (3.2x10^{-4}\,m^3/\text{sec})$$

$$= 1.7x10^{-2}\,m^3/\text{sec}$$

where:

12	=	K_{deg2} (anaerobic degradation rate for methylene chloride) from Exhibit 4-23,
2	=	d_2 (depth of solids layer) from Exhibit 4-22,
20,236	=	A (area of surface impoundment) from Exhibit 4-22,
2.5	=	Q_{sep} (rate of seepage beneath the impoundment) from Exhibit 4-22,
0.96	=	f_{d2} (fraction of total methylene chloride in solids layer that is dissolved) calculated above, and
$3.2x10^{-4}$	=	DV (rate of change in the volume of the liquid layer) calculated above.

Third, the fraction of methylene chloride reaching the solids layer that is lost to seepage is calculated using Eq. 4-126:

$$f_{sep2} = \frac{2.5\,m/yr \cdot \frac{1\,yr}{31{,}536{,}000\,sec} \cdot 0.96 \cdot 20{,}236\,m^2}{1.7x10^{-2}\,\frac{m^3}{sec}}$$

$$= 8.9x10^{-2}$$

where:

2.5	=	Q_{sep} (rate of seepage beneath the impoundment) from Exhibit 4-22,
0.96	=	f_{d2} (fraction of total methylene chloride in solids layer that is dissolved) calculated above,
20,236	=	A (area of surface impoundment) from Exhibit 4-22, and
$1.7x10^{-2}$	=	K_{tot2} (coefficient for the total rate at which methylene chloride is lost from or stored in the solids layer) calculated above.

Fourth, the fraction of each year's loading of methylene chloride lost during each year of the surface impoundment's active phase must be calculated. This calculation requires seven fractions, as shown in Eq. 4-127. From Eq. 4-122, the fraction of total methylene chloride lost from the liquid layer to volatilization is calculated:

$$f_{vol1} = \frac{5.4x10^{-6}m/sec \cdot 0.99 \cdot 20{,}236\,m^2}{0.13\,m^3/sec}$$

$$= 0.86$$

where:

$5.4x10^{-6}$	=	K_{vol1} (rate of volatilization of methylene chloride from liquid layer) calculated above,
0.99	=	f_{d1} (fraction of methylene chloride dissolved in liquid layer) calculated above,
20,236	=	A (area of surface impoundment) from Exhibit 4-22, and
0.13	=	K_{tot1} (coefficient for the total rate at which methylene chloride is lost from the liquid layer) calculated above.

Also from Eq. 4-122, the fraction of total methylene chloride lost from the liquid layer to degradation is calculated:

$$f_{deg1} = \frac{12/yr \cdot \frac{1yr}{31{,}536{,}000sec} \cdot 2m \cdot 20{,}236m^2}{0.13\frac{m^3}{sec}}$$

$$= 0.12$$

where:

12	=	K_{deg1} (anaerobic degradation rate for methylene chloride) from Exhibit 4-23,
2	=	d_l (depth of liquid layer) from Exhibit 4-22,
20,236	=	A (area of surface impoundment) from Exhibit 4-22,
0.13	=	K_{tot1} (coefficient for the total rate at which methylene chloride is lost from the liquid layer) calculated above.

Also from Eq. 4-122, the fraction of total methylene chloride lost from the liquid layer to outflow is calculated:

$$f_{out1} = \frac{2.7x10^{-4}m^3/\text{sec} \cdot 0.99}{0.13m^3/\text{sec}}$$

$$= 2.1x10^{-3}$$

where:

$2.7x10^{-4}$	=	Q_o (rate at which outflow leaves the impoundment) calculated above,
0.99	=	f_{dl} (fraction of methylene chloride dissolved in liquid layer) calculated above, and
0.13	=	K_{tot1} (coefficient for the total rate at which methylene chloride is lost from the liquid layer) calculated above.

From Eq. 4-126, the fraction of methylene chloride reaching the solids layer that is lost to degradation is calculated:

$$f_{deg2} = \frac{12/yr \cdot \frac{1yr}{31{,}536{,}000sec} \cdot 2m \cdot 20{,}236m^2}{1.7x10^{-2}\frac{m^3}{sec}}$$

$$= 0.89$$

where:

12	=	K_{deg2} (anaerobic degradation rate for methylene chloride) from Exhibit 4-23,

2	=	d_2 (depth of solids layer) from Exhibit 4-22,
20,236	=	A (area of surface impoundment) from Exhibit 4-22, and
1.7×10^{-2}	=	K_{tot2} (coefficient for the total rate at which methylene chloride is lost from or stored in the solids layer) calculated above.

The fraction of each year's loading of methylene chloride lost during each year of the surface impoundment's active phase then can be calculated from Eq. 4-127:

$$f_{act} = 0.86 + 0.12 + 2.1x10^{-3} + (1.3x10^{-2} + 2.5x10^{-3}) \cdot (0.89 + 8.9x10^{-2}) = 1.0$$

where:

0.86	=	f_{vol1} (fraction of total methylene chloride lost from liquid layer that is lost to volatilization) calculated above,
0.12	=	f_{deg1} (fraction of total methylene chloride lost from liquid layer that is lost to degradation) calculated above,
2.1×10^{-3}	=	f_{out1} (fraction of total methylene chloride lost from liquid layer that is lost in outflow from the impoundment) calculated above,
1.3×10^{-2}	=	f_{sep1} (fraction of total methylene chloride lost from liquid layer that is lost to seepage) calculated above,
2.5×10^{-3}	=	f_{del1} (fraction of total methylene chloride lost from the liquid layer as a result of the diminishing volume of the liquid layer) calculated above,
0.89	=	f_{deg2} (fraction of methylene chloride reaching the solids layer that is lost to degradation) calculated above, and
8.9×10^{-2}	=	f_{sep2} (fraction of methylene chloride reaching the solids layer that is lost to seepage) calculated above.

Fifth, the duration of the square wave is calculated by using Eq. 4-129:

$$TP = \frac{2.5x10^{8}\,\text{sec}}{1.0 \cdot 31{,}536{,}000\,\text{sec}/yr} = 8.0yr$$

where:

2.5×10^{8}	=	TF (estimated active lifetime of surface impoundment) calculated above,
1.0	=	f_{act} (fraction of each year's loading of methylene chloride lost during each year of the surface impoundment's active phase) calculated above, and
31,536,000	=	constant to convert (sec) to (yr).

Sixth, the fraction of total pollutant lost from the impoundment through seepage is calculated using Eq. 4-128:

$$f_{sep} = \frac{(1.3x10^{-2} + 2.5x10^{-3}) \cdot 8.9x10^{-2}}{1.0} = 1.4x10^{-3}$$

where:

$1.3x10^{-2}$	=	f_{sep1} (fraction of total methylene chloride lost from liquid layer that is lost to seepage) calculated above,
$2.5x10^{-3}$	=	f_{del1} (fraction of total methylene chloride lost from the liquid layer as a result of the diminishing volume of the liquid layer) calculated above,
$8.9x10^{-2}$	=	f_{sep2} (fraction of methylene chloride reaching the solids layer that is lost to seepage) calculated above, and
1.0	=	f_{act} (fraction of each year's loading of methylene chloride lost during each year of the surface impoundment's active phase).

Seventh, the average flux of methylene chloride to the unsaturated zone beneath the surface impoundment is calculated using Eq. 4-130:

$$FA_{lec,\ methylene\,chloride} = \frac{0.01\frac{(kg/ha)}{(mg/m^2)} \cdot 1.4x10^{-3} \cdot 206\frac{kg}{m^3} \cdot 4m \cdot 31.3\frac{mg}{kg}}{8.0yr}$$

$$= 4.4x10^{-2}\frac{kg}{ha-yr}$$

where:

0.01	=	constant to convert (mg/m^2) to (kg/ha),
$1.4x10^{-3}$	=	f_{sep} (fraction of total methylene chloride lost from impoundment through seepage) calculated above,
206	=	S_2 (concentration of solids in solids layer) calculated above,
4	=	d_{si} (total depth of surface impoundment) from Exhibit 4-22,
31.3	=	C_j (concentration of methylene chloride in sewage sludge) from Exhibit 4-1, and
8.0	=	TP (duration of "square wave" for approximating the loading of methylene chloride into the unsaturated soil zone) calculated above.

Eighth, Eq. 4-131 is used to calculate the average concentration of methylene chloride in seepage from the surface impoundment:

$$C_{sep,\ methylene\ chloride} = \frac{0.1\frac{(mg/L)}{(kg/ha-m)} \cdot 4.4x10^{-2}\frac{kg}{ha-yr}}{2.5\frac{m}{yr}}$$

$$= 1.7x10^{-3}\frac{mg}{L}$$

where:

0.1	=	constant to convert (kg/ha-m) to (mg/L),
$4.4x10^{-2}$	=	$FA_{lec,\ j}$ (annual average flux of methylene chloride leaching through the floor of the surface impoundment) calculated above, and
2.5	=	Q_{sep} (rate of seepage beneath the impoundment) from Exhibit 4-22.

$C_{sep,\ methylene\ chloride}$ is then used in the linked unsaturated zone and saturated zone models to estimate the concentration of methylene chloride at the well. C_{wel} for methylene chloride is modeled to be $1.4x10^{-6}$ mg/L. This concentration is then used in Eq. 4-139 to estimate human exposure:

$$EXP_{methylene\ chloride} = \frac{1.4x10^{-6}mg/L \cdot 2L/day}{70kg} = 4x10^{-8}mg/kg-day$$

where:

$1.4x10^{-6}$	=	$C_{wel,\ j}$ (concentration of methylene chloride in well water),
2	=	IW (volume of water ingested daily) from Exhibit 4-16, and
70	=	BW (body weight), assumed to be 70 kg.

Volatilization Exposure Pathway for Surface Impoundment

The volatilization of methylene chloride from a surface impoundment is calculated below. Many of the parameters required for the volatilization exposure route were calculated above for the groundwater exposure pathway, and therefore are not repeated below.

Equation 4-135 is used to calculate the fraction of total methylene chloride volatilizing during a human lifetime:

$$f_{vls} = 1.0 \cdot 0.86 = 0.86$$

where:

1.0	=	f_{act} (fraction of each year's loading of pollutant lost during each year of the surface impoundment's active phase), and
0.86	=	f_{voll} (fraction of total pollutant lost from liquid layer that is lost to volatilization) calculated above.

Equation 4-136 is then used to calculate the average flux of methylene chloride volatilizing from the surface impoundment:

$$FA_{vol,methylene\,chloride} = \frac{31.3\,mg/kg \cdot 0.86 \cdot 0.01 \cdot 206\,kg/m^3 \cdot 4\,m}{70\,yr}$$

$$= 3.2\,kg/ha\text{-}yr$$

where:

31.3	=	C_j (concentration of methylene chloride in sewage sludge) from Exhibit 4-1,
0.86	=	f_{vls} (fraction of total methylene chloride volatilizing during human lifetime) calculated above,
0.01	=	factor to convert (mg/m^2) to (kg/ha),
206	=	S_2 (concentration of solids in solids layer) calculated above,
4	=	d_{si} (total depth of surface impoundment) from Exhibit 4-22, and
70	=	LS (life expectancy), assumed to be 70 yr.

The source-receptor ratio (SRR) must be calculated next. First, the lateral virtual distance to the surface impoundment is calculated using Eq. 4-104:

$$x_y = \sqrt{\frac{20{,}236\,m^2}{\pi}} \cdot \cot\frac{22.5^\circ}{2}$$

$$= 403\,m$$

where:

20,236	=	A (area of surface impoundment) from Exhibit 4-22, and
22.5°	=	θ (the angle subtended by the surface impoundment's width) from text.

The standard deviation of the vertical distribution of concentration is calculated using Eqs. 4-105 through 4-107:

$$r' = \frac{\sqrt{20{,}236\,m^2}}{2} = 71\,m$$

where:

20,236	=	A (area of surface impoundment) from Exhibit 4-22.

From Eq. 107:

$$x = 10^{-3}\,km/m \cdot 71\,m = 0.071\,km$$

where:

10^{-3}	=	constant to convert (m) to (km), and
71	=	r' (distance from the surface impoundment's center to the receptor) calculated above.

From Eq. 106:

$$\sigma_z = \left(15.209 \cdot 0.071^{0.81558}\right) = 1.8\,m$$

where:

15.209	=	a (corresponding to x = 0.071 km) from Exhibit 4-18,
0.81558	=	b (corresponding to x = 0.071 km) from Exhibit 4-18, and
0.071	=	x (distance from the surface impoundment's center to the receptor) calculated above.

Equation 4-103 is then used to estimate the source-receptor ratio:

$$SRR = 2.032 \cdot \frac{20{,}236\,m^2 \cdot 1}{(71\,m + 403\,m) \cdot 4.5\,m/\text{sec} \cdot 1.8\,m}$$

$$= 11\,\text{sec}/m$$

where:

2.032	=	empirical constant,
20,236	=	A (area of surface impoundment) from Exhibit 4-22,
1	=	v (vertical term for dispersion of methylene chloride in air) from Exhibit 4-22,
71	=	r' (distance from the surface impoundment's center to the receptor) calculated above,
403	=	x_y (lateral virtual distance to the surface impoundment) calculated above,
4.5	=	u_{10} (average wind speed) from Exhibit 4-22, and
1.8	=	σ_z (standard deviation of the vertical distribution of methylene chloride concentration in air) calculated above.

Equation 4-137 is then used to calculate the concentration of methylene chloride in ambient air near the surface impoundment:

$$C_{air} = \frac{3.2\,\frac{kg}{ha\text{-}yr} \cdot 11\,\frac{\text{sec}}{m}}{315.36\,\frac{(kg/ha\text{-}yr)}{(\mu g/m^2\text{-sec})}}$$

$$= 0.11\,\mu g/m^3$$

where:

3.2	=	$FA_{vol,\,j}$ (annual average flux of methylene chloride volatilizing from the surface impoundment) calculated above,
11	=	SRR (source-receptor ratio) calculated above, and
315.36	=	constant to convert (kg/ha-yr) to ($\mu g/m^2$-sec).

Finally, exposure to volatilized methylene chloride is calculated using Eq. 4-138:

$$EXP_{methylene\,chloride} = \frac{10^{-3}\,mg/\mu g \cdot 0.11\,\mu g/m^3 \cdot 20\,m^3/day}{70\,kg}$$

$$= 3.1x10^{-5}\,mg/kg\text{-}day$$

where:

10^{-3}	=	constant to convert (μg) to (mg),
0.11	=	$C_{air,j}$ (average concentration of methylene chloride in ambient air at the receptor location) calculated above,
20	=	*IA* (inhalation rate) from Exhibit 4-16, and
70	=	*BW* (body weight of an adult), assumed to be 70 kg.

4.4 INCINERATION EXPOSURE METHODOLOGIES

This section evaluates human exposure to inorganic pollutants from the incineration of sewage sludge. For Round Two, the main concern is the emission of inorganic pollutants, because organic pollutant emissions were regulated as "Total Hydrocarbons" (THC) in Round One. THC was used to take into account the fact that organic pollutants are both destroyed and created in the incineration process. Although a pollutant-specific limit may be developed for dioxins and dibenzofurans in Round Two, in this Section only inorganic pollutant emissions are discussed.

The analysis uses four steps to estimate risks from incineration of sewage sludge:

1) estimate the rate at which pollutants are emitted from incinerator stacks;

2) estimate the transport and dispersion of pollutants in ambient air near incinerators, and determine the extent to which pollutant plumes overlap;

3) map expected, ground-level concentrations of pollutants onto human populations; and

4) determine the extent of human exposure to emitted pollutants and the resulting health risks.

4.4.1 Estimating Emissions of Pollutants from Incinerators

The first step in estimating human exposure to pollutants through incineration is to determine the rate at which pollutants are emitted from the stacks of sewage sludge incinerators. The rate at which an inorganic pollutant is emitted is based on the mass of pollutant entering the incinerator, the removal efficiency of the furnace, and any operating pollution control devices:

$$E_{jp} = \frac{C_j \cdot M_p \cdot (1-R_{jp})}{31{,}536{,}000 \cdot 10^3} \qquad (4\text{-}140)$$

where:

E_{jp}	=	emission rate for pollutant j at incinerator p (g pollutant/sec),
C_j	=	concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge),
M_p	=	mass of sewage sludge incinerated at incinerator p each year (kg sewage sludge/yr),
R_{jp}	=	combined removal efficiency for pollutant j of furnace and control devices for incinerator p expressed as fraction of original pollutant mass retained by the furnace or pollution control devices (dimensionless),
31,536,000	=	constant to convert (yr) to (sec), and
10^3	=	constant to convert (mg) to (g).

The rate at which a pollutant enters the incinerator is based on the feed rate for sewage sludge (M_p) and the concentration of pollutant in the sewage sludge (C_j). For a given mass of an inorganic pollutant entering the incinerator, some fraction will remain in the bottom ash of the furnace. Of the remainder, some is trapped by pollution control devices and the rest is emitted from the stack. To estimate the fraction of pollutant released to the atmosphere, the mass entering the incinerator (per unit time) is adjusted for the removal efficiency of the furnace and controls (R_{jp}). The resulting estimates for emissions from individual incinerators represent stack emissions in units of grams per second for each pollutant (E_{jp}).

4.4.2 Modeling the Dispersion of Pollutants in Air

In Round One, dispersion of pollutants in air was simulated with the Industrial Source Complex Long Term (ISCLT) model (Bowers et al., 1980; U.S. EPA, 1986) as implemented in the Graphical Exposure Modeling System for personal computers, or PC-GEMS (U.S. EPA, 1989a). The model described the dispersion of pollutants as steady-state Gaussian plumes, and allowed the user several modeling options.

In Round One, all incinerator stacks were modeled as point sources. Depending on the velocity and temperature of exit gases, plume rise was modeled as either momentum- or buoyancy-induced; the appropriate option was selected automatically by the program. Both the downwash and plume-rise-by-distance options were used, but (for lack of sufficient data) the effects of surrounding terrain were ignored. For computational efficiency, the dispersion of pollutants near each incinerator was modeled only once, using a unit rate of emissions (i.e., one g/sec of pollutant emitted per kg/sec of sewage sludge incinerated). Resulting dispersion estimates were converted to ground-level concentrations at individual locations, scaled by expected emissions of each pollutant from each individual incinerator.

4.4.3 Mapping Dispersion and Pollutant Concentrations Onto a Unified Grid

In Round One, results from the ISCLT model were reported as dispersion ratios in units of $\mu g/m^3$ of pollutant concentration in ambient air per g/sec of pollutant emissions from incinerator stacks. Separate coefficients were provided for selected locations in the area surrounding an individual incinerator. The model allowed the user to choose between a rectangular or polar grid for specifying these locations. In Round One, the rectangular grid was selected and coordinates specified in such a way that results from the modeling of individual incinerators could be integrated into a unified mapping of dispersion ratios for the U.S. as a whole. Explicit details of this analysis are provided in the Risk Assessment Document (U.S. EPA, 1993a).

Within each cell of the grid system, expected pollutant concentrations were calculated by combining emission estimates from each incinerator with results from ISCLT. When a cell was

impacted by more than one incinerator, pollutant concentrations were summed to calculate total expected concentrations for that cell:

$$AA_{ij} = \sum_{p=1}^{n} D_{ip} \cdot E_{jp} \tag{4-141}$$

where:

AA_{ij}	=	estimated ambient air concentration of pollutant j in grid cell i due to sewage sludge incineration (μg pollutant/m^3 air),
D_{ip}	=	dispersion ratio for grid cell i impacted by incinerator p (μg pollutant/m^3 air per g pollutant/sec), and
n	=	number of incinerators modeled.

For Round Two, rather than model the entire U.S. for this Comprehensive Hazard Identification exercise, modeling results from the analyses performed for Round One were used. For each of the 172 sewage sludge incinerators modeled in Round One, the maximum dispersion ratio for any cell impacted by the incinerator was identified. The geometric mean of these maximum dispersion ratios was then calculated. (An arithmetic average was not calculated because the maximum dispersion ratios appeared to be log-normally distributed.) The geometric mean of the incinerator's sewage sludge feed rates was also determined; again, the feed rates appeared to be log-normally distributed. Using an average removal efficiency for any given pollutant, an "average" ambient air concentration was estimated:

$$AA_j = D_{avg} \cdot E_{avg,\,j} \quad or$$

$$AA_j = D_{avg} \cdot \left(\frac{C_j \cdot M_{avg} \cdot (1-R_j)}{31{,}536{,}000 \cdot 10^3} \right) \tag{4-142}$$

where:

AA_j	=	average ambient air concentration of pollutant j due to sewage sludge incineration (μg pollutant/m^3 air),
D_{avg}	=	geometric mean of the maximum dispersion ratios for 172 incinerators (μg pollutant/m^3 air per g pollutant/sec),
$E_{avg,j}$	=	average emission rate for pollutant j (g pollutant/sec),
C_j	=	concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge),
M_{avg}	=	geometric mean of sewage sludge feed rates for 172 incinerators (kg sewage sludge/yr),
R_j	=	combined removal efficiency for pollutant j of furnace and control devices expressed as fraction of original pollutant mass retained by the furnace or pollution control devices (dimensionless),
31,536,000	=	constant to convert (yr) to (sec), and
10^3	=	constant to convert (mg) to (g).

4.4.4 Estimating Human Exposure

Once average ambient air concentrations of each inorganic pollutant are estimated, an estimate of human exposure is made by combining the concentrations with assumptions about daily inhalation volume and body weight. Individual exposure to each pollutant is calculated as:

$$EXP_j = \frac{AA_j \; IA \; 10^{-3}}{BW} \tag{4-143}$$

where:

EXP_j	=	exposure to pollutant j (mg pollutant/kg body weight-day),
IA	=	inhalation rate (m^3 air/day),
10^{-3}	=	constant to convert (μg) to (mg), and
BW	=	body weight (kg).

As can be seen from the equation, the conservative assumption that each person inhales air at the estimated (outdoor) concentration for 24 hours per day for his or her entire lifetime is made. It is also assumed that all of the inhaled pollutant is absorbed into the body, and thus exposure is effectively equivalent to dose.

4.4.5 Data Inputs

To estimate human exposure through this pathway, several types of incinerator data are required: sewage sludge feed rates, dispersion ratios, and removal efficiencies. As mentioned in Section 4.4.3, average values for these parameters were calculated. For sewage sludge feed rates, a geometric mean of the feed rates for the 172 sewage sludge incinerators was calculated to be 1.04×10^6 kg/yr. For dispersion ratios, after the maximum dispersion ratio had been identified for each incinerator, the geometric mean was calculated to be 3.36 $\mu g/m^3$ per g/sec.

The removal efficiencies in two types of incinerators, multiple hearth with wet scrubber and fluidized bed with wet scrubber, were needed for inorganic pollutants. Unfortunately, removal efficiency data were not available for any of the inorganics other than beryllium, which was already evaluated in Round One. Therefore, exposures were calculated for two removal efficiencies: 50 percent and 90 percent. Fifty percent was chosen as a very conservative number; 90 percent was chosen as a more reasonable number, based on the removal efficiencies of inorganic pollutants evaluated in Round One.

4.4.6 Example Calculations for Incineration

To estimate human exposure to manganese from an incinerator with 50 percent removal efficiency, Eq. 4-142 is used first to estimate the average ambient air concentration of manganese:

$$AA_{manganese} = \frac{3.36\ (\mu g/m^3)\ /\ (g/\text{sec}) \cdot 1620\ mg/kg \cdot 1.04x10^6\ kg/yr \cdot (1-0.5)}{31{,}536{,}000\ \text{sec}/yr \cdot 10^3\ mg/g}$$

$$= 0.090\ \mu g/m^3$$

where:

3.36	=	D_{avg} (geometric mean of maximum dispersion ratios) from text,
1620	=	C_j (manganese concentration in sewage sludge) from Exhibit 4-1,
1.04×10^6	=	M_{avg} (geometric mean of sewage sludge feed rates) from text,
0.5	=	R_j (combined removal efficiency) assumed to be 50 percent,
31,536,000	=	constant to convert (yr) to (sec), and
10^3	=	constant to convert (mg) to (g).

To then calculate the exposure to manganese, Eq. 4-143 is used:

$$EXP_{manganese} = \frac{0.090\ \mu g/m^3 \cdot 20\ m^3/day \cdot 10^{-3}\ mg/\mu g}{70\ kg} = 2.6x10^{-5}\ mg/kg\text{-}day$$

where:

0.090	=	AA_j (average ambient air concentration of manganese) calculated above,
20	=	IA (inhalation rate) from Exhibit 4-16,
10^{-3}	=	constant to convert (μg) to (mg), and
70	=	BW (body weight), assumed to be 70 kg.

4.5 RISK CALCULATIONS

In Sections 4.2, 4.3, and 4.4, estimates were made of exposure to pollutants from sewage sludge that is land applied, placed in surface disposal sites, or incinerated. In this section, the exposure estimates are combined with human and ecological toxicity values to obtain estimates of risk by pollutant and exposure pathway. If risk values are greater than certain thresholds for a given exposure pathway, that pathway is defined as "critical" for that pollutant.

4.5.1 Human Health Risk Calculations

For land application Pathways 1, 2, 4, 5, 12, 13, 14, and 15, and for surface disposal and incineration, human exposure estimates to carcinogenic pollutants are combined with cancer potency slopes to estimate individual risk:

$$CI_j = EXP_j \cdot q_j^* \tag{4-144}$$

where:

CI_j = incremental cancer risk from pollutant j for exposed individual (incremental risk of developing cancer per lifetime of exposure),

EXP_j = exposure to pollutant j (mg pollutant/kg body weight-day), and

q_j^* = human cancer potency of pollutant j (mg pollutant/kg body weight-day)$^{-1}$.

For land application Pathway 3, the exposure must be modified to account for the duration of exposure relative to lifetime:

$$CI_j = EXP_j \cdot DE \cdot q_j^* \tag{4-145}$$

where:

DE = exposure duration adjustment (number of years of exposure divided by expected lifetime of 70 years).

For Pathway 3, an exposure duration adjustment of (5/70) was used.

In this analysis, if the individual risk for a given pollutant and exposure pathway exceeded a value of 10^{-4} (one in 10,000), then the pathway was considered critical.

For non-carcinogenic pollutants for land application Pathways 1 through 5 and 12 through 15, as well as for surface disposal and incineration, estimated exposure was compared to the Risk Reference Dose (RfD):

$$RNC_j = \frac{EXP_j}{RfD_j} \tag{4-146}$$

where:

RNC_j = ratio of the exposure to the *RfD* for pollutant *j* (dimensionless), and

RfD_j = risk reference dose for pollutant *j* (mg pollutant/kg body weight-day).

If the ratio was equal to or greater than one for a given pollutant and exposure pathway, then the pathway was considered critical.

Since May, 1993, when human toxicity data were first obtained for the 31 pollutant candidates, some pollutant toxicity numbers have been changed, and others withdrawn. Exhibit 4-24 presents the q_1^* and RfD values used in the Comphrehensive Hazard Identification for both oral and inhalation exposure routes and indicates those numbers which have changed from those presented in Exhibit 3-7.

EXHIBIT 4-24
Human Health Toxicity Numbers[1]

Pollutant	Cancer Potency Slopes (risk/mg/kg-day)		Reference Doses (mg/kg-day)	
	Inhalation	Oral	Inhalation	Oral
Acetic acid (2,4-dichlorophenoxy)		1.9×10^{-2}		1.0×10^{-2}
Aluminum				
Antimony				4.0×10^{-4}
Asbestos[3]				
Barium			1.4×10^{-4} (2)	7.0×10^{-2}
Beryllium	8.4	4.3		5.0×10^{-3}
Bis(2-ethylhexyl)phthalate		1.4×10^{-2}		2.0×10^{-2}
Boron			5.7×10^{-3}	9.0×10^{-2}
Butanone, 2-			8.6×10^{-2}	6.0×10^{-1} (2)
Carbon Disulfide			2.9×10^{-3}	1×10^{-1}
Cresol, p-				(3)
Cyanides (soluble salts and complexes)				2.0×10^{-2}
Dioxins and dibenzofurans	3.0×10^{6} (4)	3.0×10^{6} (4)		
Endosulfan-II				5.0×10^{-5}

EXHIBIT 4-24
Human Health Toxicity Numbers (cont'd)[1]

Pollutant	Cancer Potency Slopes (risk/mg/kg-day)		Reference Doses (mg/kg-day)	
	Inhalation	Oral	Inhalation	Oral
Fluoride				6.0×10^{-2}
Manganese			5×10^{-5}(2)	5.0×10^{-3}(2), (5)
Methylene chloride	1.5×10^{-3}	7.5×10^{-3}	9.0×10^{-1}	6.0×10^{-2}
Nitrate				1.6
Nitrite				1.0×10^{-1}
Pentachloronitrobenzene		2.6×10^{-1}		3.0×10^{-3}
Phenol				6.0×10^{-1}
Polychlorinated biphenyls -- coplanar		7.7		
Propanone, 2-				1.0×10^{-1}
Propionic acid, 2-(2,4,5-trichlorophenoxy)				8.0×10^{-3}
Silver				5.0×10^{-3}
Thallium				8.0×10^{-5}(2)
Tin				6.0×10^{-1}
Titanium				(3)
Toluene			4×10^{-1}(2)	2.0×10^{-1}
Trichlorophenoxyacetic acid, 2,4,5-				1.0×10^{-2}
Vanadium				7.0×10^{-3}

Notes:
[1] See Section 3.3.4 for a complete description of sources reviewed; see Exhibit 3-7 for references for individual toxicity numbers unless marked with a (2).
[2] Toxicity data from IRIS (March 3, 1995).
[3] Toxicity number withdrawn from IRIS for further consideration (March 3, 1995). No toxicity number available in HEAST (March, 1994 tables).
[4] U.S. EPA (1994c).
[5] This RfD is for water intake, assuming the necessary amount of the trace nutrient has already been ingested with food.

For land application Pathway 11, exposure was compared to an occupational Threshold Limit Value-Time Weighted Average (TLV-TWA). The TLV-TWA is the time-weighted average concentration of a pollutant to which nearly all workers can be repeatedly exposed over an 8-hour workday and 40-hour work week, without adverse effect. The ratio of exposure to the TLV-TWA was taken:

$$RTC_j = \frac{EXP_j}{TLV\text{-}TWA_j} \tag{4-147}$$

where:

RTC_j = ratio of the exposure to the TLV-TWA for pollutant j (dimensionless), and

$TLV\text{-}TWA_j$ = Threshold Limit Value-Time Weighted Average for pollutant j (mg pollutant/m^3 air).

Exhibit 4-25 presents the Threshold Limit Values used in this analysis. As shown in the Exhibit, values were not available for three organic compounds.

EXHIBIT 4-25
Threshold Limit Values for Pollutants

Pollutant	Threshold Limit Values-Time Weighted Average (mg/m^3)[1]
Acetic acid (2,4-dichlorophenoxy)	10
Aluminum	2[2]
Antimony	0.5
Asbestos	0.2 fiber/cm^3 [3]
Barium	0.5
Beryllium	0.002
Bis(2-ethylhexyl) phthalate	5
Boron	1[4]
Butanone, 2-	590
Carbon disulfide	31[5]
Cresol, p-	22[5]
Cyanides (soluble salts and complexes)	10[5,6]

Pollutant	Threshold Limit Values-Time Weighted Average (mg/m^3)[1]
Dioxins and Dibenzofurans	
Endosulfan-II	0.1[5]
Fluoride	1.6
Manganese	1[7]
Methylene chloride	174
Nitrate	NA
Nitrite	NA
Pentachloronitrobenzene	0.5[5]
Phenol	19[5]
Polychlorinated biphenyls (coplanar)	0.5[5]
Propanone, 2-	1780
Propionic acid, 2-(2,4,5-trichlorophenoxy)	
Silver	0.01[8]
Thallium	0.1[5]
Tin	2
Titanium	10[9]
Toluene	188
Trichlorophenoxy acetic acid, 2,4,5-	
Vanadium	0.05

Notes:
NA means Not Applicable.
[1] American Conference of Governmental Industrial Hygienists (1994).
[2] Soluble salts.
[3] Crocidolite.
[4] Anhydrous sodium tetraborate.
[5] Skin.
[6] Hydrogen cyanide.
[7] Fume.
[8] Soluble compounds.
[9] Titanium dioxide.

Example Calculations for Human Risk

For carcinogens, this example estimates the risk associated with the application of sewage sludge containing dioxins and dibenzofurans to agricultural land. The exposure pathway is Pathway 12, ingestion of fish and water from surface water that receives eroded soil. In the example calculation for exposure presented in Section 4.2.12, the sum of the exposures to dioxins in water and fish was estimated to be $5x10^{-11}$ mg/kg-day. To estimate the individual risk from this exposure to these carcinogens, Eq. 4-144 is used:

$$CI_{dioxins} = 5.2x10^{-11}\frac{mg}{kg-day} \cdot 3.0x10^{6}\frac{risk}{mg/kg-day}$$

$$= 2x10^{-4}$$

where:

$5.2x10^{-11}$	=	EXP_j (exposure to dioxins) calculated in Section 4.2.12, and
$3.0x10^{6}$	=	q^*_j (human cancer potency of dioxins) from Exhibit 4-24.

Given that the individual risk exceeds 10^{-4}, this pathway is critical for dioxins and dibenzofurans.

To demonstrate the use of an exposure duration adjustment for carcinogens, exposure to a carcinogen must be calculated for Pathway 3. Using Eq. 4-5 for beryllium in sewage sludge used on agricultural land:

$$EXP_{beryllium} = \frac{\frac{0.2g}{day} \cdot \frac{10^{-3}kg}{g} \cdot \frac{8mg}{kg}}{16kg}$$

$$= 1.0x10^{-4}\frac{mg}{kg-day}$$

where:

0.2	=	*IS* (sewage sludge ingestion rate for agricultural land) from Section 4.2.3,
10^{-3}	=	constant to convert (g) to (kg),
8	=	C_j (concentration of beryllium in sewage sludge) from Exhibit 4-1, and
16	=	*BW* (body weight of child assumed to be exposed to agricultural land) from Section 4.2.3.

Then using Eq. 4-145:

$$CI_{beryllium} = 1.0x10^{-4}\frac{mg}{kg-day} \cdot \frac{5}{70} \cdot 4.3\frac{risk}{(mg/kg-day)}$$

$$= 3x10^{-5}$$

where:

$1.0x10^{-4}$	=	EXP_j (exposure to beryllium) calculated above,
5/70	=	DE (exposure duration adjustment) from text above, and
4.3	=	q^*_j (human cancer potency of beryllium) from Exhibit 4-24.

Given that the individual risk does not exceed 10^{-4}, this pathway is not critical for beryllium.

For non-carcinogens, boron in sewage sludge applied to forest lands in Pathway 4 provides an example. Total dietary exposure to boron from wild animals was calculated in Section 4.2.4 to be 0.029 mg/kg-day. Using Eq. 4-146:

$$RNC_{boron} = \frac{0.029\frac{mg}{kg-day}}{0.090\frac{mg}{kg-day}} = 0.3$$

where:

0.029	=	EXP_j (exposure to boron) calculated in section 4.2.4, and
0.090	=	RfD_j (risk reference dose for boron) from Exhibit 4-24.

This ratio of exposure to RfD is less than one, and thus this pathway is not critical for boron.

For Pathway 11, a different type of risk to human health is calculated. Using the exposure to manganese calculated in Section 4.2.11 and using Eq. 4-147:

$$RTC_{manganese} = \frac{4.1x10^{-3}\frac{mg}{m^3}}{1\frac{mg}{m^3}} = 4.1x10^{-3}$$

where:

$4.1x10^{-3}$	=	EXP_j (exposure to manganese) calculated in section 4.2.11, and

1 = $TLV\text{-}TWA_j$ (Threshold Limit Value-Time Weighted Average for manganese) from Exhibit 4-25.

This ratio of exposure to *TLV-TWA* is less than one, and thus this pathway is not critical for manganese.

4.5.2 Ecological Risk Calculations

For Pathways 6, 7, 9, and 10, to estimate risk to an herbivorous or an insectivorous mammal, or an earthworm, an ecological risk quotient was calculated. The ecological risk quotient is the ratio of the predicted exposure to an appropriate toxicological reference value:

$$RQ_j = \frac{EXPA_j}{TRV_j} \tag{4-148}$$

where:

RQ_j = ecological risk quotient for pollutant j (dimensionless),
$EXPA_j$ = exposure of animal to pollutant j (mg pollutant/kg diet), and
TRV_j = toxicological reference value for pollutant j for an animal (mg pollutant/kg diet).

Ideally, for toxicological reference values (*TRV*s), data for livestock, earthworms, and small mammals such as shrews and moles would be available; however, toxicity data are generally not available for all of these species. Instead, toxicity tests are most often performed on a select number of "laboratory species," such as rats, mice, and dogs. Results from tests on these species are assumed to be representative of the sensitivity of species experiencing exposure in the field.

Toxicological reference values for livestock and small mammals were taken from three different sources: ATSDR (Agency for Toxic Substances and Disease Registry) documents, WHO (World Health Organization) documents, and data provided in HSDB (Hazardous Substances Data Bank). Each of these sources summarized results of toxicity studies. For this analysis, none of the original studies were obtained. The documents reported either NOAELs (no observed adverse effect levels), LOAELs (lowest observed adverse effect levels), or information necessary to calculate NOAELs or LOAELs. When only a LOAEL was provided, the LOAEL was divided by a factor of ten to make it more comparable with the NOAELs. For a given pollutant, the lowest NOAEL (or adjusted LOAEL value) reported in any of the sources was chosen to be the toxicological reference value for that species. This conservative practice was deemed appropriate for this effort.

As shown in Eq. 4-8 in Section 4.2.6, animal exposure is calculated in terms of concentration of pollutants in the food items (mg/kg); therefore, the toxicological reference values also should be in concentration units (mg/kg). While HSDB provides more details on the protocol of the toxicity studies, the ATSDR and WHO documents only report the toxicity test

results in terms of mg/kg-day, regardless of the exposure metric actually employed in the toxicity tests. When the reference values came from sources where the exposures were given as mg pollutant/kg body weight-day doses, the values were converted to a mg/kg food concentration. In many of the toxicity studies, exposure was probably originally reported as food concentrations. However, because the original studies were not consulted, standard values were used for the data needed to make the conversions from the mg pollutant/kg body weight-day values reported in the ATSDR and WHO documents and HSDB to food concentration equivalents, as described below.

To make the conversions from mg pollutant/kg body weight-day to food concentration, data on body weights, food consumption rates, and, sometimes, water consumption rates were needed. Body weights for the various species in the tests were taken from the table of reference body weights in EPA's report *Recommendations for and Documentation of Biological Values for Use in Risk Assessment* (Table 1-2, U.S. EPA, 1988). Allometric equations for daily food and water consumption were taken from Table 1-3 of the same source. Daily doses were converted to food concentrations by multiplying the dose by the body weight and dividing by the daily food consumption rate.

When the exposure was given in terms of pollutant concentration in drinking water, the water concentration was first converted to a mg pollutant/kg body weight-day dose by multiplying by the drinking water rate and dividing by the body weight. The dose was then converted to a food concentration in the same manner as described in the previous paragraph. In essence, the water concentration was multiplied by the ratio of the water consumption rate to the food consumption rate.

Exhibit 4-26 presents the *TRVs* used for each pollutant for Pathways 6, 7, and 10. It also shows which species was used to derive the *TRV*, whether exposure conversions were necessary, and whether the *TRV* was based on a NOAEL or a LOAEL. Toxicological reference values could not be obtained for endosulfan-II or 2-propanone.

EXHIBIT 4-26
Toxicological Reference Values for Mammals

Pollutant	Toxicological Reference Value (mg pollutant/kg diet)	Reference (Species)
Acetic acid (2,4-dichlorophenoxy)[1]	180	HSDB (mammals)[N]
Aluminum[1]	1400	Domingo et al., 1987 (rat)[N]
Antimony[1]	0.34	Schroeder et al., 1970 (rat)[L]
Barium[1]	0.70	Perry et al., 1983, 1985, 1989 (rat)[N]
Beryllium[1]	9.1	Schroeder and Mitchener, 1975 (rat)[N]

EXHIBIT 4-26
Toxicological Reference Values for Mammals (cont.'d)

Pollutant	Toxicological Reference Value (mg pollutant/kg diet)	Reference (Species)
Bis(2-ethylhexyl) phthalate[1]	130	Ganning et al., 1991 (rat)[N]
Boron[1]	190	Weir and Fisher, 1972 (dog)[N]
Butanone, 2-[1]	2300	Ralston et al., 1985 (rat)[N]
Carbon disulfide[1]	80	Jones-Price et al., 1984 (rabbit)[L]
Cresol, p-[1]	410	Hornshaw et al., 1986 (mink)[N]
Cyanides (soluble salts and complexes)[1]	10	Gerhart, 1987 (rat)[N]
Dioxins and Dibenzofurans	1.3×10^{-5}	Kociba et al., 1978 (rat)[N]; Murray et al., 1979 (rat)[L]
Fluoride[2]	11	HSDB (mice)[N]
Manganese[1]	17	Laskey et al., 1982 (rat)[L]
Methylene chloride[1]	78	Serota et al., 1986 (rat)[N]
Pentachloronitrobenzene[3]	40	HSDB (dog)[L]
Phenol[1]	8200	NCI, 1980 (rat)[N]
Polychlorinated biphenyls (coplanar)[1]	0.13	Barsotti and Van Miller, 1984 (monkey)[N]
Propionic acid, 2-(2,4,5-trichlorophenoxy)	30	HSDB (dog)[N]
Silver[1]	9.9	Rungby and Danscher, 1984 (mouse)[L]
Thallium[1]	5.2	Downs et al., 1960 (rat)[N]
Tin[1]	0.91	Schroeder et al., 1968 (rat)[L]
Titanium[1]	27	Schroeder et al., 1964 (mouse)[N]
Toluene[1]	4000	NTP, 1990 (rat)[N]
Trichlorophenoxy acetic acid, 2,4,5-[1]	39	HSDB (rat)[N]
Vanadium[1]	3.9	Domingo et al., 1985 (rat)[N]

Notes:
[1] Secondary source reported exposure in units of mg pollutant/kg body weight/day.

[2] Secondary source reported exposure in units of mg pollutant/kg drinking water.
[3] Secondary source reported exposure in units of mg pollutant/kg food.
[N] Toxicological Reference Value was based on a NOAEL.
[L] Toxicological Reference Value was based on a LOAEL.

For soil-dwelling organisms in Pathway 9, *TRVs* also were needed. The soil-dwelling biota includes a taxonomically very diverse array of organisms. There are very few toxicological data available, however, for most groups of soil-dwelling organisms. In ecological risk assessments, one or a few species of earthworms are generally chosen to represent the soil-dwelling niche. Therefore, searches focused on toxicity data for earthworms for the Round Two candidate pollutants.

For Pathway 9, exposure to pollutants in soil by earthworms is measured by the concentration of the pollutants in the sewage sludge/soil; therefore, the *TRVs* for earthworms should be in units of soil concentration. While there is a considerable amount of toxicity data for earthworms, the variability in test quality and designs makes results difficult to compare; thus, it is very difficult to assess potential environmental hazards of pollutants to earthworms (Roberts and Dorough, 1985; Edwards and Bohlen, 1992). There have been recent attempts to standardize earthworm toxicity testing protocols (Roberts and Dorough, 1984; Callahan et al., 1994). However, while standard laboratory testing protocols should help in comparing toxicities of different pollutants or sensitivities of different species of earthworms, many of the laboratory testing protocols produce results that are almost impossible to interpret in terms of field exposure (Edwards and Bohlen, 1992). For instance, while placing earthworms in contact with pollutants on filter papers in Petri dishes for two days may provide measurable LC_{50}s, it is not clear how these results can be used to determine risk from exposure to pollutants in the field. In short, there is a dearth of toxicity data for earthworms that can be used in risk assessments.

The toxicity literature was searched through computerized databases (e.g., BIOSIS and HSDB) and recent review articles were examined to find toxicity information on earthworms or other soil biota for the Round Two candidate pollutants (e.g., Callahan et al., 1994; Edwards and Bohlen, 1992; Beyer, 1990; Roberts and Dorough, 1985). Usable data were found only for two of the pollutants. For phenol, Neuhauser and Callahan (1990) determined a NOAEC (no observed adverse effect concentration) of 5900 mg pollutant/kg soil for mortality in the earthworm *Eisenia fetida*. The earthworms were exposed for eight weeks to phenol in a combination of sand and horse manure in Petri dishes. In a separate study, Hartenstein et al. (1981) investigated the effects of cations (including heavy metals) and anions added to activated sewage sludge on the growth of *E. fetida*. The only Round Two metal investigated was manganese. It was determined that manganese at the highest level tested (22,000 mg pollutant/kg sludge) was innocuous to this species of earthworms. Therefore, a conservative NOAEC estimate of 22,000 mg pollutant/kg sludge is used for manganese. Exhibit 4-27 summarizes these available data.

EXHIBIT 4-27
Toxicological Reference Values for Soil-Dwelling Organisms

Pollutant	Toxicological Reference Value (NOAEC in mg pollutant/kg sewage sludge)	Reference (Species)
Manganese	22,000	Hartenstein et al., 1981 (*Eisenia fetida*)
Phenol	5,900	Neuhauser and Callahan, 1990 (*Eisenia fetida*)

Example Calculation for Ecological Risk

To estimate risk to animals from sewage sludge-amended land, the same equation, Eq. 4-148, is used for Pathways 6, 7, 9, and 10. As an example, the risk for predators of soil-dwelling organisms (Pathway 10) through exposure to manganese in sewage sludge-amended agricultural soil can be estimated using Eq. 4-148:

$$RQ_{manganese} = \frac{10\,mg/kg}{17\,mg/kg} = 0.6$$

where:

10 = $EXPA_j$ (exposure to manganese) calculated in Section 4.2.10, and
17 = TRV_j (toxicological reference value for manganese) from Exhibit 4-26.

This ratio of exposure to *TRV* is less than one, and thus agricultural Pathway 10 is not a critical pathway for manganese.

4.5.3 Human Health and Ecological Risk Results

Presented in this section are estimates of risk for those pollutant-exposure pathway combinations for which all pollutant-specific data are available. For these estimates, 95th percentile pollutant concentrations, with non-detects set equal to the minimum detection level, as determined in the 1988 National Sewage Sludge Survey were used. Risks for each of the land application exposure pathways are presented first, followed by risks for surface disposal and incineration. Note that in the following exhibits, a blank entry means that a risk estimate could not be calculated for that pollutant-pathway combination, either because not all pollutant-specific data were available or, in the case of human endpoint pathways, because neither q_1* or *RfD* values were available.

EXHIBIT 4-28
Risk Results for Highly Exposed Individual for Pathway 1

Pollutant	Individual Cancer Risk				Exposure as a Fraction of RfD			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Dioxins and Dibenzofurans	9×10^{-6}							
Fluoride					0.1			

EXHIBIT 4-29
Risk Results for Highly Exposed Individual for Pathway 2

Pollutant	Individual Cancer Risk	Exposure as a Fraction of RfD
	Agricultural Land	Agricultural Land
Dioxins and Dibenzofurans	3×10^{-5}	
Fluoride		0.5

EXHIBIT 4-30
Risk Results for Highly Exposed Individual for Pathway 3

Pollutant	Individual Cancer Risk				Exposure as a Fraction of RfD			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Acetic acid (2,4-dichlorophenoxy)	5×10^{-10}	2×10^{-10}	2×10^{-10}	5×10^{-10}	<0.01	<0.01	<0.01	<0.01
Antimony					0.8	0.6	0.6	0.8
Barium					0.3	0.3	0.3	0.3
Beryllium	3×10^{-5}	1×10^{-5}	1×10^{-5}	3×10^{-5}	0.02	0.02	0.02	0.02
Bis(2-ethylhexyl) phthalate	2×10^{-6}	8×10^{-7}	8×10^{-7}	2×10^{-6}	0.1	0.1	0.1	0.1
Boron					0.03	0.02	0.02	0.03
Butanone, 2-					<0.01	<0.01	<0.01	<0.01
Carbon disulfide					<0.01	<0.01	<0.01	<0.01
Cyanides (soluble salts and complexes)					0.08	0.07	0.07	0.08
Dioxins and Dibenzofurans	8×10^{-4}	3×10^{-4}	3×10^{-4}	8×10^{-4}				
Endosulfan-II					0.02	0.01	0.01	0.02
Fluoride					0.09	0.07	0.07	0.09
Manganese					4	3	3	4
Methylene chloride	2×10^{-7}	7×10^{-8}	7×10^{-8}	2×10^{-7}	<0.01	<0.01	<0.01	<0.01
Pentachloronitrobenzene	2×10^{-8}	6×10^{-9}	6×10^{-9}	2×10^{-8}	<0.01	<0.01	<0.01	<0.01
Phenol					<0.01	<0.01	<0.01	<0.01

EXHIBIT 4-30
Risk Results for Highly Exposed Individual for Pathway 3 (cont'd.)

Pollutant	Individual Cancer Risk				Exposure as a Fraction of RfD			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Polychlorinated biphenyls (coplanar)	4×10^{-5}	1×10^{-5}	1×10^{-5}	4×10^{-5}				
Propanone, 2-					0.02	0.01	0.01	0.02
Propionic acid, 2-(2,4,5-trichlorophenoxy)					<0.01	<0.01	<0.01	<0.01
Silver					0.3	0.3	0.3	0.3
Thallium					2	1	1	2
Tin					<0.01	<0.01	<0.01	<0.01
Toluene					0.02	0.01	0.01	0.02
Trichlorophenoxy acetic acid, 2,4,5-					<0.01	<0.01	<0.01	<0.01
Vanadium					0.1	0.1	0.1	0.1

EXHIBIT 4-31
Risk Results for Highly Exposed Individual for Pathway 4

Pollutant	Individual Cancer Risk			Exposure as a Fraction of RfD		
	Agricultural Land	Forest	Reclamation Site	Agricultural Land	Forest	Reclamation Site
Boron					0.3	0.1
Manganese					10	40
Polychlorinated biphenyls (coplanar)	$2x10^{-3}$	$6x10^{-2}$	$2x10^{-3}$			

EXHIBIT 4-32
Risk Results for Highly Exposed Individual for Pathway 5

Pollutant	Individual Cancer Risk			Exposure as a Fraction of RfD		
	Agricultural Land	Forest	Reclamation Site	Agricultural Land	Forest	Reclamation Site
Barium				<0.01	<0.01	<0.01
Boron				<0.01	<0.01	<0.01
Manganese				<0.01	<0.01	<0.01
Polychlorinated biphenyls (coplanar)	$1x10^{-4}$	$1x10^{-4}$	$1x10^{-4}$			

EXHIBIT 4-33
Risk Results for Highly Exposed Individual for Pathway 6

Pollutant	Risk Quotient[1]			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Aluminum	100	80	100	80
Beryllium	0.04	0.3	0.03	0.3
Boron	0.7	4	0.7	4
Dioxins and Dibenzofurans	0.03	0.6	0.03	0.6
Fluoride	10	30	10	30
Manganese	200	800	200	800
Polychlorinated biphenyls (coplanar)	5	80	4	80
Silver	0.02	0.3	0.01	0.3
Titanium	7	0.7	7	0.7

[1] Ratio of exposure to ecological toxicological reference value.

EXHIBIT 4-34
Risk Results for Highly Exposed Individual for Pathway 7

Pollutant	Risk Quotient[1]		
	Agricultural Land	Forest	Reclamation Site
Acetic acid (2,4-dichlorophenoxy)	<0.01	<0.01	<0.01
Aluminum	0.4	0.4	0.4
Antimony	1	1	1
Barium	40	40	40
Beryllium	0.01	0.01	0.01
Bis(2-ethylhexyl) phthalate	0.02	0.02	0.02
Boron	0.01	0.01	0.01
Butanone, 2-	<0.01	<0.01	<0.01
Carbon disulfide	<0.01	<0.01	<0.01
Cresol, p-	0.01	0.01	0.01
Cyanides (soluble salts and complexes)	0.2	0.2	0.2
Dioxins and Dibenzofurans	0.4	0.4	0.4
Fluoride	0.6	0.6	0.6
Manganese	1	1	1
Methylene chloride	<0.01	<0.01	<0.01
Pentachloronitrobenzene	<0.01	<0.01	<0.01
Phenol	<0.01	<0.01	<0.01
Polychlorinated biphenyls (coplanar)	0.6	0.6	0.6
Propionic acid, 2-(2,4,5-trichlorophenoxy)	<0.01	<0.01	<0.01
Silver	0.2	0.2	0.2
Thallium	0.03	0.03	0.03
Tin	2	2	2

EXHIBIT 4-34
Risk Results for Highly Exposed Individual for Pathway 7 (cont'd.)

Pollutant	Risk Quotient[1]		
	Agricultural Land	Forest	Reclamation Site
Titanium	0.2	0.2	0.2
Toluene	<0.01	<0.01	<0.01
Trichlorophenoxy acetic acid, 2,4,5-	<0.01	<0.01	<0.01
Vanadium	0.2	0.2	0.2

[1] Ratio of exposure to ecological toxicological reference value.

EXHIBIT 4-35
Risk Results for Highly Exposed Individual for Pathway 9

Pollutant	Risk Quotient [1]			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Manganese	0.02	0.07	0.02	0.07
Phenol	<0.01	<0.01	<0.01	<0.01

[1] Ratio of exposure to ecological toxicological reference value.

EXHIBIT 4-36
Risk Results for Highly Exposed Individual for Pathway 10

Pollutant	Risk Quotient[1]			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Antimony	0.2	3	0.2	3
Barium	10	50	10	50
Dioxins and Dibenzofurans	4	80	3	80
Fluoride	5	8	5	8
Manganese	0.6	2	0.6	2
Thallium	<0.01	0.04	<0.01	0.04

[1] Ratio of exposure to ecological toxicological reference value.

EXHIBIT 4-37
Risk Results for Highly Exposed Individual for Pathway 11

Pollutant	Exposure as a Fraction of TLV-TWA
Aluminum	0.1
Antimony	<0.01
Barium	<0.01
Beryllium	<0.01
Bis(2-ethylhexyl) phthalate	<0.01
Boron	<0.01
Butanone, 2-	<0.01
Cresol, p-	<0.01
Cyanides (soluble salts and complexes)	<0.01
Endosulfan-II	<0.01
Fluoride	<0.01
Manganese	<0.01
Methylene chloride	<0.01
Phenol	<0.01
Polychlorinated biphenyls (coplanar)	<0.01
Propanone, 2-	<0.01
Silver	<0.01
Thallium	<0.01
Tin	<0.01
Titanium	<0.01
Toluene	<0.01
Vanadium	<0.01

EXHIBIT 4-38
Risk Results for Highly Exposed Individual for Pathway 12

Pollutant	Individual Cancer Risk				Exposure as a Fraction of RfD			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Acetic acid (2,4-dichlorophenoxy)	3×10^{-16}	8×10^{-17}	3×10^{-16}	4×10^{-17}	<0.01	<0.01	<0.01	<0.01
Antimony					<0.01	<0.01	<0.01	<0.01
Bis(2-ethylhexyl) phthalate	1×10^{-9}	3×10^{-10}	9×10^{-10}	1×10^{-10}	<0.01	<0.01	<0.01	<0.01
Butanone, 2-					<0.01	<0.01	<0.01	<0.01
Carbon disulfide					<0.01	<0.01	<0.01	<0.01
Cyanides (soluble salts and complexes)					<0.01	<0.01	<0.01	<0.01
Dioxins and Dibenzofurans	2×10^{-4}	2×10^{-4}	7×10^{-5}	2×10^{-4}				
Endosulfan-II					<0.01	<0.01	<0.01	<0.01
Methylene chloride	2×10^{-17}	6×10^{-18}	2×10^{-17}	3×10^{-18}	<0.01	<0.01	<0.01	<0.01
Pentachloronitrobenzene	3×10^{-12}	8×10^{-13}	3×10^{-12}	3×10^{-13}	<0.01	<0.01	<0.01	<0.01
Phenol					<0.01	<0.01	<0.01	<0.01
Polychlorinated biphenyls (coplanar)	2×10^{-6}	3×10^{-6}	1×10^{-6}	3×10^{-6}				
Propanone, 2-					<0.01	<0.01	<0.01	<0.01
Propionic acid, 2-(2,4,5-trichlorophenoxy)					<0.01	<0.01	<0.01	<0.01
Silver					<0.01	<0.01	<0.01	<0.01
Toluene					<0.01	<0.01	<0.01	<0.01
Trichlorophenoxy acetic acid, 2,4,5-					<0.01	<0.01	<0.01	<0.01

EXHIBIT 4-39
Risk Results for Highly Exposed Individual for Pathway 13

Pollutant	Individual Cancer Risk				Exposure as a Fraction of RfD			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Butanone, 2-					<0.01	<0.01	<0.01	<0.01
Carbon disulfide					<0.01	<0.01	<0.01	<0.01
Dioxins and Dibenzofurans	1×10^{-4}	5×10^{-4}	1×10^{-3}	3×10^{-4}				
Methylene chloride	9×10^{-9}	3×10^{-8}	9×10^{-8}	2×10^{-8}	<0.01	<0.01	<0.01	<0.01
Polychlorinated Biphenyls (coplanar)	2×10^{-6}	1×10^{-5}	3×10^{-5}	7×10^{-6}				
Toluene					<0.01	<0.01	<0.01	<0.01

EXHIBIT 4-40
Risk Results for Highly Exposed Individual for Pathway 14

Pollutant	Individual Cancer Risk				Exposure as a Fraction of RfD			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Acetic acid (2,4-dichlorophenoxy)	0	0	0	0	0	0	0	0
Antimony					20	40	3	60
Barium					9	20	1	20
Beryllium	$7x10^{-4}$	$9x10^{-4}$	$3x10^{-4}$	$1x10^{-3}$	0.03	0.04	0.01	0.05
Bis(2-ethylhexyl) phthalate					0	0	0	0
Butanone, 2-					<0.01	<0.01	<0.01	<0.01
Carbon disulfide					<0.01	<0.01	<0.01	<0.01
Cyanides (soluble salts and complexes)					<0.01	<0.01	<0.01	<0.01
Dioxins and Dibenzofurans	0	0	0	0				
Endosulfan-II					0	0	0	0
Fluoride					0.2	0.3	0.08	0.4
Manganese					700	1000	30	2000
Methylene chloride	$5x10^{-12}$	$1x10^{-11}$	$5x10^{-12}$	$1x10^{-11}$	<0.01	<0.01	<0.01	<0.01
Pentachloronitrobenzene	0	0	0	0	0	0	0	0
Phenol					0	0	0	0
Polychlorinated biphenyls (coplanar)	0	0	0	0				

EXHIBIT 4-40
Risk Results for Highly Exposed Individual for Pathway 14 (cont'd.)

Pollutant	Individual Cancer Risk				Exposure as a Fraction of RfD			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Propanone, 2-					0	<0.01	0	<0.01
Propionic acid, 2-(2,4,5-trichlorophenoxy)					0	0	0	0
Silver					<0.01	<0.01	<0.01	<0.01
Toluene					0	0	0	0
Trichlorophenoxy acetic acid, 2,4,5-					0	0	0	0
Vanadium					0.2	0.3	0.1	0.4

EXHIBIT 4-41
Risk Results for Highly Exposed Individual for Pathway 15

Pollutant	Individual Cancer Risk -- Scenario 2[1]			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Dioxins and Dibenzofurans	9×10^{-3}	4×10^{-2}	9×10^{-3}	5×10^{-3}
Polychlorinated biphenyls (coplanar)	3×10^{-4}	7×10^{-3}	2×10^{-4}	1×10^{-6}

[1] Note that at this time individual cancer risks can only be calculated for Scenario 2. Scenario 1 is for non-cancer health effects.

EXHIBIT 4-42
Risk Results for Highly Exposed Individual for the Surface Disposal Pathways

Pollutant	Monofills				Surface Impoundments			
	Groundwater Pathway		Air Pathway		Groundwater Pathway		Air Pathway	
	Ind. Cancer Risk	Exposure as a Fraction of RfD	Ind. Cancer Risk	Exposure as a Fraction of RfD	Ind. Cancer Risk	Exposure as a Fraction of RfD	Ind. Cancer Risk	Exposure as a Fraction of RfD
Antimony		NE				4		
Barium		NE				1		
Beryllium	NE	NE			2×10^{-4}	0.01		
Manganese		NE				90		
Dioxins and Dibenzofurans	NE		NE		3×10^{-11}		7×10^{-4}	

NE = Not evaluated in the Comprehensive Hazard Identification. See Section 4.3.6 for explanation.

EXHIBIT 4-43
Risk Results for Highly Exposed Individual for Incineration Pathway

Pollutant	Exposure/RfD for 50% Removal Efficiency	Exposure/RfD for 90% Removal Efficiency
Barium	0.2	0.04
Boron	<0.01	<0.01
Manganese	0.5	0.1