Risk Assessment
Technical Background Document for the Chlorinated Aliphatics Listing Determination
Addendum

September 29, 2000

USEPA Office of Solid Waste
Economics, Methods, and Risk Analysis Division

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1. INTRODUCTION

On August 25, 1999, the Environmental Protection Agency (EPA) published a proposed rulemaking for wastewaters and wastewater treatment sludges from the production of chlorinated aliphatic chemicals. In conjunction with this proposed rulemaking EPA prepared a "Risk Assessment Technical Background Document for the Chlorinated Aliphatics Listing Determination" (with attached Addendum) dated July 30, 1999 ("1999 Risk Assessment TBD"; USEPA 1999).

The purpose of this 2000 Addendum is 1) to provide information that EPA prepared in response to public and peer review comment on the proposed rulemaking that modifies, clarifies, or supplements information we presented in the 1999 Risk Assessment TBD and 2) to present a list of errata for the 1999 Risk Assessment TBD. This supplemental background document is intended as a companion to the 1999 Risk Assessment TBD, and is not intended to be a stand-alone document[1]. Sections 2 through 6 of this Addendum generally correspond to Sections 2 through 6 of the 1999 Risk Assessment TBD, and discuss issues by the following topic areas:

- Establishing contaminant exposure scenarios (Section 2)
- Estimating exposure point concentrations (Section 3)
- Exposure and toxicity assessments (Section 4)
- Risk characterization (Section 5)

Section 6 provides references for this addendum. Section 7 provides new versions of Appendices D.1, D.2, D.4, D.5, and F.1 that have been modified in response to peer review comments. Section 8 provides a list of errata for the 1999 Risk Assessment TBD.

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[1] The terminology used in this document is the same as that used in the 1999 Risk Assessment TBD and is not redefined in this Addendum.

2. ESTABLISHING CONTAMINANT EXPOSURE SCENARIOS

This section provides technical information that supplements that provided in Section 2 of the 1999 Risk Assessment TBD. Section 2 of the background document discussed how EPA established the exposure scenarios that we evaluated in the risk assessment. An exposure scenario describes how an individual (a receptor) may come into contact with (be exposed to) contaminants in a waste.

The information we provide in this section includes:

- Support for EPA's assumption that beef and dairy cattle might plausibly be raised in the vicinity of chlorinated aliphatics manufacturing facilities;
- An evaluation of our assumptions regarding the modeled pasture size; and
- A discussion of how certain exposure assumptions influence our probabilistic risk estimates.

2.1 Support for EPA's Assumptions Regarding the Plausibility of Beef and Dairy Cattle Farming in the Vicinity of Chlorinated Aliphatics Facilities

Commenters on the proposed rule questioned EPA's assumption that a farmer might raise fruits, exposed vegetables, root vegetables, beef cattle and dairy cows in the vicinity of chlorinated aliphatics production facilities. The commenters confirmed that it was plausible to assume that beef cattle might be raised in the vicinity of chlorinated aliphatics facilities that manage wastewaters; however, they questioned our assumptions regarding the presence of dairy cattle in the parts of the country where chlorinated aliphatics facilities are located. To respond to this concern, EPA reviewed publicly available data from the 1997 agricultural census (http://govinfo.library.orst.edu/ag-stateis.html) that are summarized in Table 2-1. As shown in Table 2-1, EPA determined that of the 23 chlorinated aliphatics production facilities,

Table 2-1. Presence of Beef Cattle and Dairy Cows in Counties Where Chlorinated Aliphatics Production Facilities are Located

Facility and Location	County/Parish	Beef		Dairy	
		Farms	Cows	Farms	Cows
GE, Waterford, NY	Saratoga	98	1225	79	6446
FMC, Baltimore, MD	Baltimore (county)	149	2210	31	2865
Formosa, Baton Rouge, LA	East Baton Rouge	255	10901	13	791
Borden, Geismar, LA Vulcan, Geismar, LA	Ascension	189	5913	6	21
Occidental, Convent, LA	St. James	10	NR	0	0
Dow, Plaquemine, LA Georgia Gulf, Plaquemine, LA	Iberville	78	NR	2	NR
Westlake, Calvert City, KY	Marshall	298	6020	9	134
Dow Corning, Midland, MI	Midland	87	1011	13	704
Occidental, Deer Park, TX Geon, LaPorte, TX	Harris	983	28590	31	844
Occidental (Oxymar), Gregory, TX	San Patricio	210	NR	2	NR
Dow, Freeport, TX	Brazoria	1178	50452	31	56
Formosa, Point Comfort, TX	Calhoun	163	NR	1	NR
PPG, Lake Charles, LA Condea Vista, Westlake, LA	Calcasieu	518	25140	14	23
Dow Corning, Carrolton, KY	Carroll	174	4716	10	221
DuPont/Dow, Louisville, KY	Jefferson	170	NR	4	NR
Velsicol, Memphis, TN	Shelby	229	4980	10	42
DuPont/Dow, LaPlace, LA	St. John the Baptist	8	NR	1	NR
Shell, Norco, LA	St. Charles	50	2723	0	0
Vulcan, Wichita, KS	Sedgwick	419	10692	63	4797

NR = not reported

21 facilities are located in counties with at least one farm that reported having dairy cows (beef cattle were reported in all counties), confirming that ingestion of contaminated home-produced dairy products is a plausible exposure scenario.

EPA collected additional information for the land treatment unit. First, in 1997 dairy cattle were reported for the parish (county) in which the land treatment unit is located. Furthermore, a land use map for the area surrounding the facility shows that cropland and pastureland bounds the facility (Figure 2-1). Lastly, a 1986 RCRA facility Assessment Report (RFA) confirms at least the historical proximity of cattle to waste management units at the facility (Figure 2-2).

2.2 Evaluation of EPA's Assumptions Regarding the Modeled Pasture Size

In the 1999 Risk Assessment Technical Background Document we presented an analysis of risk for an adult farmer and a child of a farmer. We explained that our evaluation considered that the farmer and his/her child resides on an agricultural farm with dimensions of approximately 1410 meters (m) by 1410 m, and consumes fruits, exposed vegetables, and root vegetables grown on the farm, as well as beef and dairy products produced from cattle raised on the farm. We assumed the farm was used to provide forage, grain, and silage for the cattle. We estimated that high end risk would exceed 1×10^{-5} (the hazardous waste listing program's target risk level) for dioxin under the aerated biological treatment tank and land treatment unit waste management scenarios. Commenters on the proposed rule questioned the plausibility our assumptions regarding the productivity of the farm that we evaluated. They suggested that the exposure scenario that EPA evaluated is unrealistic in that it implies that a farm not only has both a dairy and beef cattle operation, but raises grain and silage, in addition to crops for human consumption, while still maintaining enough pasture to graze the animals.

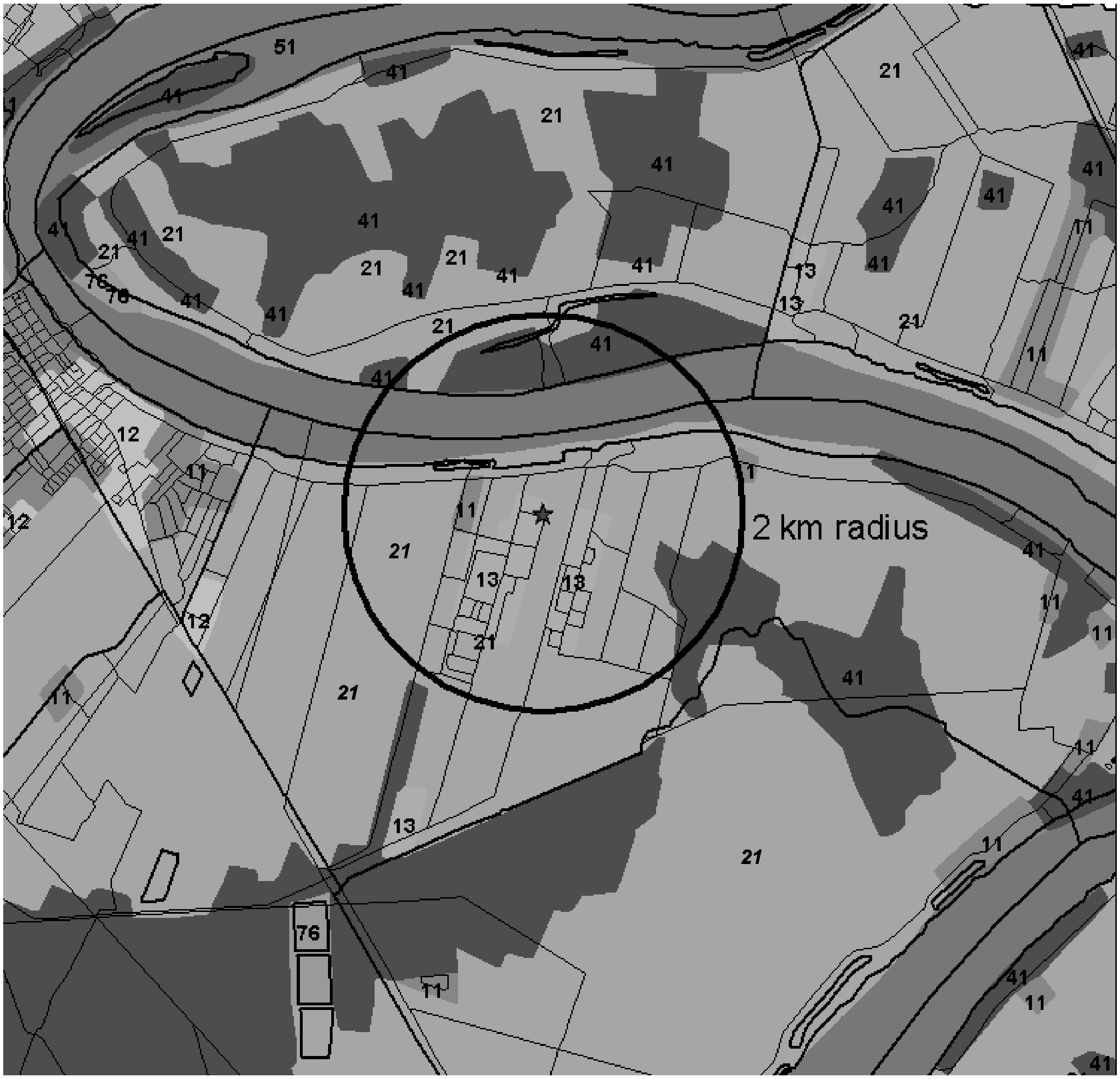


Figure 2-1. Land Use Near the Land Treatment Unit

Code	Land Use Description	Code	Land Use Description
11	Residential	41	Deciduous Forest Land
12	Commercial and Services	51	Streams and Canals
13	Industrial	76	Transitional Areas
21	Cropland and Pasture		

Figure 2-2. Excerpt from 1986 Land Treatment Unit RFA

Georgia Gulf
Plaquemine, LA
12/86 RFA

A. UNIT 8 SUMMARY

1. UNIT NAME: Landfarm/Land Treatment Area.

2. UNIT DESCRIPTION: The site is approximately 300 acres and is located in the southern, undeveloped section of the Georgia Gulf facility [16].

   Currently, solids are loaded by a front-end loader into a dump truck for transport to the land farm area six times a day. A dozer is used to level the solids and incorporate them into the soil [17].

3. DATE OF START-UP: Mid-1970's [16].

4. DATE OF CLOSURE: Unknown

5. WASTES MANAGED: Lime solids generated from a cold lime water softening process, brine solids generated from brine purification [17], and wastes from normal maintenance and cleaning of earthen ditches throughout the entire complex [8].

6. RELEASE CONTROLS: Unknown

7. HISTORY OF RELEASES: Unknown; however, the State issued a violation to the facility for allowing cows to graze in this area.

B. CONCLUSIONS: The materials being handled seem to indicate there is a very low potential for an air release from this facility. Further information is needed to assess the potential for a surface water release. The soil/ground release potential from this unit is quite high. Various file correspondence indicated that the wastes that are being placed in this area were analyzed except for ditch cleaning materials. However, the file review did not locate any results from the testing.

C. FURTHER ACTION: Obtain information, by a letter of inquiry, on the history and operational procedures for this unit. The results of analyses on the lime and brine solids should be obtained to further discuss soil/groundwater and surface water releases. Depending on the information obtained, it may be necessary for a site visit to obtain soil samples from this unit.

Tables 5-3 and 5-8 of the 1999 Risk Assessment Technical Background Document show that for the adult farmer 1) 99.3 percent of the high end dioxin risk for chlorinated aliphatic wastewaters is due to ingestion of beef and dairy products, and only 0.7 percent is due to ingestion of home grown fruits and vegetables, and 2) 96 percent of the high end dioxin risk for wastewater treatment sludges managed in a land treatment unit is due to ingestion of beef and dairy products, and only 4 percent is due to ingestion of home grown fruits and vegetables. As a result, even though EPA believes it is plausible that a subsistence or hobby farmer would raise fruits and vegetables for home consumption, the validity of EPA's risk estimate depends almost entirely on the validity of our assumption that a farmer might consume both beef and dairy products from cattle raised on a farm located in the vicinity of a chlorinated aliphatics production facility. Moreover, under the tank scenario the dioxins in the beef and dairy products result almost exclusively from the cattle's intake of forage that is contaminated by air emissions from the modeled wastewater treatment tank – negligible levels of dioxins are contributed to cattle as a result of the cattle's ingestion of grain, silage, or soil (see Appendix H.1, Table H.1-1a of the 1999 Risk Assessment Technical Background Document). Similarly, under the land treatment unit scenario the dioxins in the beef and dairy products result primarily from the cattle's intake of forage and soil that are contaminated by air emissions and runoff/erosion from the modeled land treatment unit – minor levels of dioxins are contributed to cattle as a result of the cattle's ingestion of grain or silage (see Table 5-1). Consequently, all that is required for the adult farmer to realize the risk that EPA presented in the proposed rule is that the farmer consume beef and dairy products derived from cattle that consume forage and, in the case of the land treatment unit, incidentally ingest soil, from the farmer's pastureland/field. That is, it is not necessary that we assume that the farmer consumes home-grown fruits and vegetables, or that the farmer produces grain or silage for use as cattle feed.

In response to the commenters' concerns, we reviewed our methodology for estimating the concentrations of dioxins in forage and soil to ensure that we were adequately considering the size of the contaminated pasture (field) versus its expected productivity. For aerated biological treatment tanks we evaluated risks due to vapor air emissions from the tanks. For the land treatment unit, we evaluated risks due to vapor air emissions, particulate air emissions, and erosion/runoff emissions from the land treatment unit. In the proposed rule we explained that in evaluating the air pathway we always assume that the cattle are located along the centerline of the area most greatly impacted by air releases from the waste management units (64 FR 46486). We said that for the wastewater tank analysis, the air concentrations within about a 100-meter lateral distance from this point did not vary appreciably, and stated specifically in our 1999 Risk Assessment Technical Background Document Addendum that the concentrations varied about 20% within 200 meters of the point of maximum concentration. In the course of our reevaluation of these data in response to comments, we felt that we likely should have considered how the concentrations of dioxins in air, and therefore in forage, vary over a wider aerial extent that would be more consistent with the area of a pasture.

We concluded that a pasture that is approximately 275m by 275m (75,625m^2, approximately 19 acres) likely would be large enough to support sufficient cattle to sustain the family of a subsistence farmer. We originally based our estimate of pasture size on a University of Kentucky research pasture located in Kentucky's Bluegrass region (two of the chlorinated aliphatics facilities are located in the Bluegrass region of Kentucky) (www.bae.uky.edu/~lturner/gps/udalres.htm). This pasture was 225m X 275m in size. We contacted a researcher at the University of Kentucky who told us that eight cattle graze on the pasture nearly year-around (Turner, 1999). We later supplemented our information by contacting a Professor and Extension Dairy Specialist with the Texas A&M University Animal Science Extension Program Unit (Jordan, 2000). We were told that under normal climate conditions in the Gulf Coast region of Texas, approximately 4 to 5 acres of pasture per dairy cow would be required under a subsistence farming scenario, whereas with intensive grazing management, a farmer could achieve <1 acre per dairy cow in the same region. Consequently, we estimate that in the areas of the country where chlorinated aliphatics facilities are located, 19 acres of pasture could sustain, at a minimum, 4 or 5 cows (beef cows consume less forage than dairy cows, therefore should require somewhat less pasture).

We estimated how many cows would be necessary to sustain a family of four as follows:

	Reference	Dairy		Beef	
		HE	CT	HE	CT
Daily Consumption Rate of Dairy and Beef (pre-cooked) – Adult Farmer, kg*/day	USEPA (1999), Table K-2	2.1	0.73	0.3234	0.098
Daily Consumption of ***Home-Produced*** Dairy and Beef – Adult Farmer, kg*/day	Previous row multiplied by 0.25 for dairy and 0.49 for beef (USEPA, 1999, Section 2.2.1 and Table K-2)	0.525	0.1825	0.048	0.1585
Consumption of ***Home-Produced*** Dairy and Beef for four-person Family kg*/day kg*/year	(Conservatively assumes that all family members are adults.) Previous row multiplied by 4 Previous row multiplied by 365	2.1 766	0.73 266	0.634 231	0.192 70
Average Production per Cow	Dairy: Texas A&M (undated), average for Texas Beef: USEPA (1999), Section 5.2	17 kg/day 6218 kg/year		338 kg (dressed weight of 1 steer)	
Number of Cows Required to Sustain four-person Family	Consumption of Home-Produced Dairy/Beef Divided by Average Dairy/Beef Production per Cow	1		1 per year	

* wet or fresh weight

Based on this calculation, we estimate that a farmer would require a minimum three to four cows to sustain a family of four. One dairy cow would provide 6205 kg/year of milk, compared to the required 766 kg/year (based on the high end consumption rate); however, two dairy cows are necessary to ensure a continuous supply of milk (Jordan, 2000) since one or the other cow will be dry periodically. One beef cow would provide 338 kg of beef, which is more than the necessary 231 kg required by the family of four each year (based on the high end consumption rate); however, a farmer likely would be raising one or two additional beef cattle for future consumption.

We used the results of the air modeling we conducted for the proposed rulemaking to determine the approximate difference between the air concentration that we used to calculate the proposed risk estimate (the air concentration corresponding to a point located 300m from the modeled wastewater treatment tank and land treatment unit) and the average air concentration at a 75,625m^2 field located 300m from the modeled wastewater treatment tank and land treatment unit. The model we used to evaluate the erosion/runoff pathway under the land treatment unit exposure scenario assumed that contaminants are deposited evenly across the entire (1410m by 1410m) modeled agricultural field (farm). Consequently, for the erosion/runoff pathway under the land treatment unit scenario, we would have reasonably evaluated a pasture that would be a portion of this farm.

Our analysis focuses exclusively on how modifying the pasture size influences the vapor phase concentrations, since the vapor phase concentration would most influence the risk estimate. Further, we were only concerned with the "dry deposition" of vapors, since, as stated in USEPA, 1998: "... wet deposition of vapor-phase lipophilic compounds [such as dioxins] can be considered negligible." Our analysis also did not include evaluation of the impact of increasing pasture size on the deposition of particulates. This is because we did not evaluate particulate emissions from wastewater treatment tanks, and because dioxin risks due to particle-phase air emissions under the land treatment unit scenario were much less than those due to the vapor-phase air emissions which dominated the air pathway dioxin risk estimate. As an example, the high end vapor-phase air pathway risk for the adult farmer, 1E-04, was 98 percent of the total air pathway risk estimate, compared to the particle-phase air pathway risk, 2E-06, which was the remaining 2 percent (see Table 5-1).

The sections below describe the steps in the methodology we used to evaluate how a larger pasture size would influence our estimates of vapor phase concentration, therefore risk, under the aerated biological wastewater treatment tank and the EDC/VCM sludge land treatment unit waste management scenarios. We did not evaluate how a larger pasture size influences the results of any of our landfill scenario risk estimates, since the air pathway risks for the landfill scenarios were already below levels of concern.

1. Identify the data sources. As explained in Appendix D.3 of the 1999 Risk Assessment Technical Background Document, we use the ISC model to generate "unitized" air concentrations at various "receptor locations" that result from the dispersion of emissions from a source. The receptor locations in the air dispersion modeling runs are the points, established on a grid, at which air concentrations are estimated. (These receptors should not be confused with the use of the term "receptors" to describe the types of individuals [for example, resident, farmer, gardener] who might be exposed to the contaminants we are evaluating.) We multiply the unitized air concentrations by contaminant-specific emissions estimates to produce contaminant-specific air concentrations. In conducting the air dispersion modeling, receptors are placed around the source at fixed distances. We identify the maximum unitized air concentrations to define the centerline of the contaminant plume.

For the tank scenario, the data we used as the basis for our modifications to our pasture size assumptions were the air dispersion modeling data on which our high end deterministic risk estimates for the adult farmer were based: the data set corresponding to the central tendency meteorological location (Memphis, TN) and the central tendency tank size (approximately 14m by 14m) (the high end parameters for the adult farmer scenario were waste concentration and exposure duration). For the land treatment unit we based our pasture size modification on the air dispersion modeling data for the meteorological station that corresponds to the single land treatment unit location, Baton Rouge, and single land treatment unit area, 687,990m^2.

2. Determine the receptor locations that define the field. To estimate the average air concentration over a 275m by 275m area that begins 300m from the source, we had to identify the unitized air concentrations at receptors

- Located between a distance 300m from the source (the distance to receptor in the high end analyses) and a distance 575m (300m plus 275m) from the source and
- Located within 137.5m (275m÷2) of each side of the plume centerline.

Neither the tank dispersion modeling nor the land treatment unit dispersion modeling was performed with receptors placed exactly at points that would define the edges of a 275m by 275m field. The modeling for the tank was conducted for receptors placed 300m, 400m, 500m, and 750m from the source, and the modeling for the land treatment unit was conducted for receptors placed 300m, 400m, 500m, and 1000m from the source. Moreover, the lateral distance between receptors varied with distance from the source, as shown in Tables 2-2 and 2-3. Fortunately, the nature of the air dispersion modeling allowed us to interpolate the unitized air concentrations at the field edges from the available data (see Figure 2-3 for the tank and

Figure 2-4 for the land treatment unit). The shaded concentrations shown in Tables 2-4 and 2-5 present the data matrices (the air dispersion modeling output) from which the concentrations at the lateral field edges were interpolated.

Table 2-2. Distance of Receptor Points from Source and Distance between Receptors – Tank

Distance from Source (m)	*Distance between Receptors (m)*
300	38.7
400	50.9
500	63.7
750	94.6

Table 2-3. Distance of Receptor Points from Source and Distance between Receptors – Land Treatment Unit

Distance from Source (m)	*Distance between Receptors (m)*
300	178.7
400	203.7
500	228.7
1000	353.7

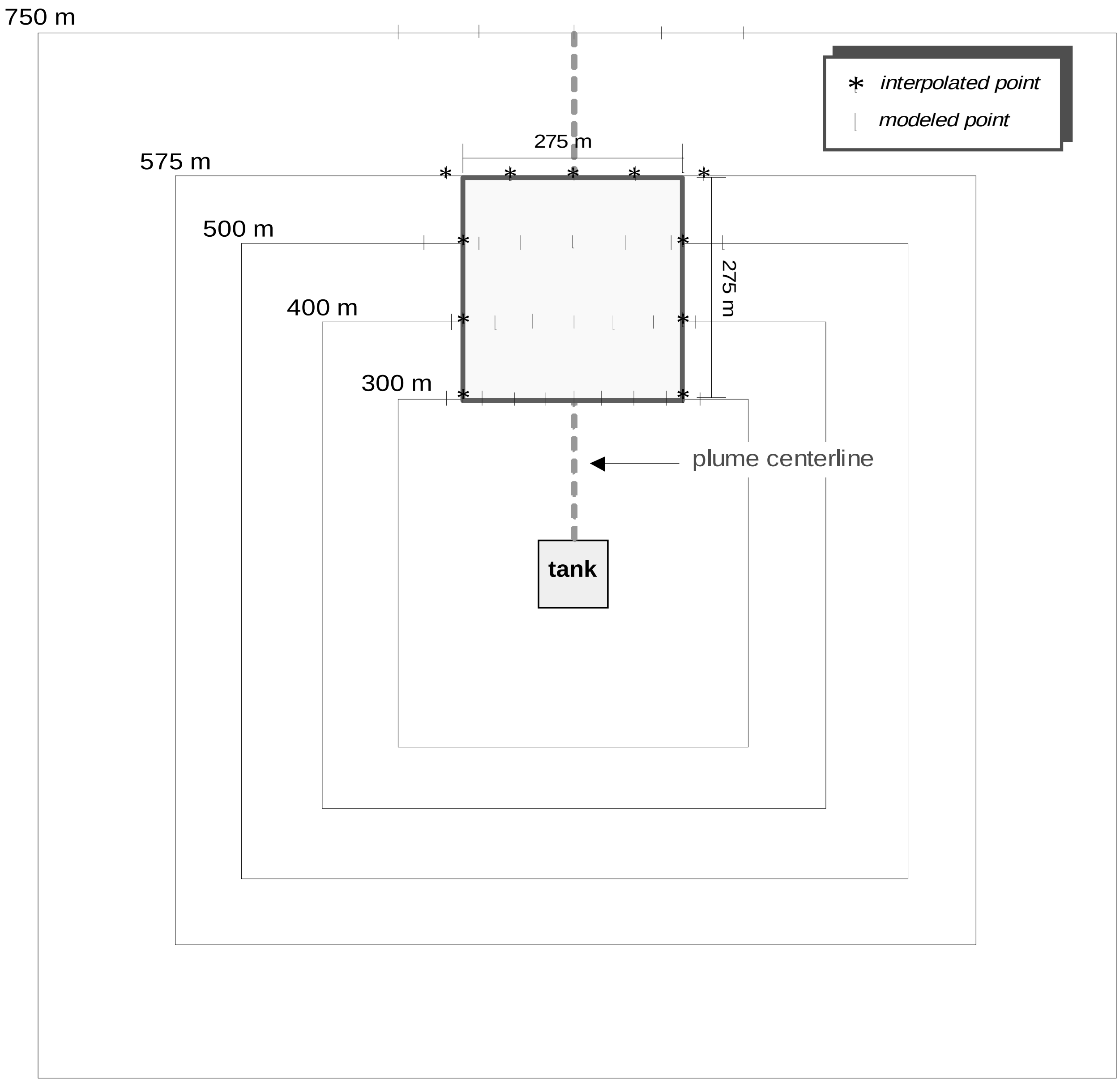


Figure 2-3. Air Modeling Scenario for 275 by 275 m Pasture Located 300 m from the Edge of the Wastewater Tank

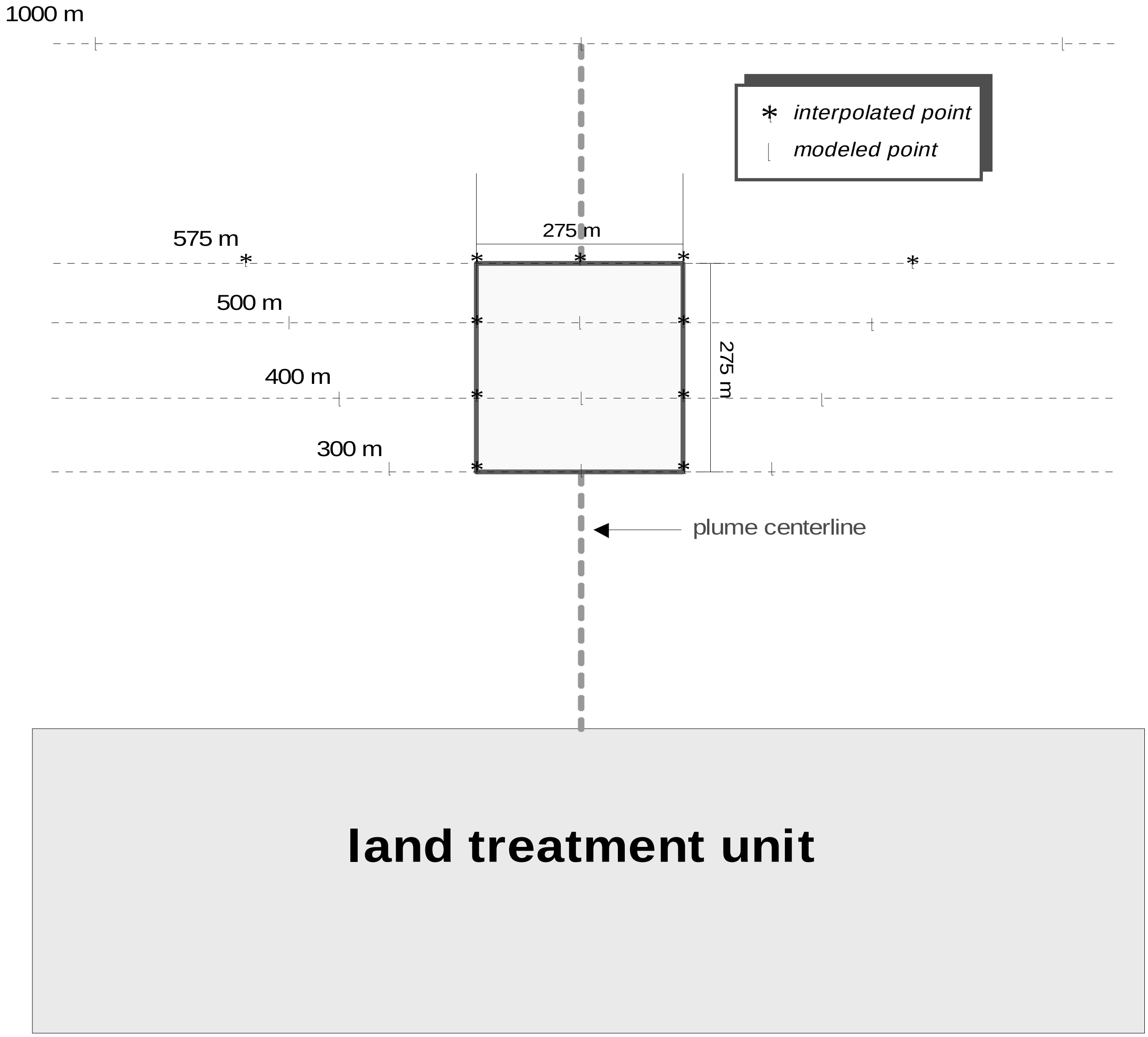


Figure 2-4. Air Modeling Scenario for 275 by 275 m Pasture Located 300 m from the Edge of the Land Treatment Unit

Table 2-4. Tank, Memphis, TN - Unitized Air Concentrations

Receptor Direction	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)
	300 m	**400 m**	**500 m**	**750 m**
SW	1.2568	0.77034	0.52422	0.26347
	1.36036	0.83322	0.56593	0.28168
	1.5424	0.94746	0.64469	0.32066
	1.84392	1.13388	0.7719	0.38329
SSW	2.32404	1.44046	0.98551	0.49226
	2.70777	1.68476	1.15532	0.57808
	2.58606	1.60202	1.09507	0.54391
	2.60314	1.61774	1.10832	0.55223
S	2.63624	1.64077	1.12551	0.56196
	2.00194	1.23111	0.83705	0.41168
	1.73581	1.07005	0.72866	0.35958
	1.50264	0.92407	0.62874	0.31029
SSE	1.25577	0.77024	0.52305	0.25772
	1.01697	0.62045	0.42023	0.20661
	0.92195	0.56353	0.3824	0.18897
	0.82953	0.50616	0.34313	0.16981
SE	0.72703	0.44181	0.29898	0.14847
	0.86244	0.5265	0.35701	0.17684
	1.02141	0.62602	0.42539	0.21077
	1.07255	0.65529	0.44408	0.21837
ESE	1.18654	0.7293	0.49609	0.24515
	1.32842	0.82322	0.56272	0.27987
	1.10624	0.6794	0.46154	0.2269
	1.2783	0.79413	0.54326	0.2697
E	1.28475	0.79691	0.5448	0.27007
	1.23271	0.76335	0.52091	0.25754
	1.17376	0.72208	0.49077	0.24169
	1.52736	0.9438	0.64406	0.3198
ENE	1.86402	1.15047	0.78464	0.38987
	1.90276	1.17311	0.79972	0.39807
	1.66429	1.01668	0.68888	0.34022
	1.68193	1.03258	0.70236	0.35094
NE	1.57969	0.96892	0.65975	0.33224
	1.79242	1.09862	0.74673	0.37238
	2.13602	1.31674	0.89816	0.44888
	2.37193	1.46154	0.99638	0.49604

(continued)

Table 2-4. Tank, Memphis, TN - Unitized Air Concentrations

Receptor Direction	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)
	300 m	**400 m**	**500 m**	**750 m**
NNE	2.62736	1.63238	1.11892	0.56127
	2.40257	1.48303	1.01158	0.50177
	2.46014	1.51459	1.03091	0.50842
	3.26319	2.0155	1.37468	0.67938
N	4.70998	2.94723	2.02793	1.01646
centerline	5.13137	3.2088	2.20578	1.10428
	5.06794	3.1763	2.18709	1.09957
	4.3813	2.74817	1.89414	0.9559
NNW	3.18078	1.96877	1.34505	0.6709
	2.85208	1.77628	1.21914	0.6146
	2.20599	1.35984	0.9271	0.4635
	1.94446	1.20143	0.82073	0.41383
NW	1.60932	0.9919	0.67762	0.34373
	1.60011	0.98095	0.66656	0.33271
	1.97633	1.22909	0.84323	0.42639
	2.05037	1.27335	0.87209	0.43827
WNW	2.00156	1.25075	0.85977	0.43334
	1.62487	1.00492	0.68577	0.34049
	2.03714	1.27086	0.87229	0.43628
	2.63879	1.64879	1.13252	0.56596
W	3.65328	2.30719	1.59568	0.8057
	2.48129	1.53589	1.0482	0.51902
	2.20659	1.38011	0.94865	0.47549
	1.76247	1.09148	0.74597	0.3711
WSW	1.63511	1.00919	0.68861	0.3428
	1.66917	1.03496	0.70879	0.35605
	1.48373	0.9108	0.61941	0.30803
	1.43385	0.88224	0.60111	0.30107

Table 2-5. Land Treatment Unit, Baton Rouge, LA - Unitized Air Concentrations

Receptor Direction	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)	Concentration (ug/m3 / mg/s-m2)
	300 m	**400 m**	**500 m**	**1000 m**
	3.47154	2.60508	2.04333	0.89154
WSW	3.13568	2.40502	1.92803	0.92179
	2.72803	2.12778	1.72687	0.83396
SW	2.18539	1.7153	1.39844	0.66837
	2.70771	2.08753	1.68106	0.78631
SSW	3.33091	2.50543	1.96314	0.88249
centerline	3.49363	2.71231	2.1852	1.00329
S	3.26376	2.52376	2.03361	0.97445
	2.76422	2.10757	1.68424	0.77
SSE	2.06589	1.50596	1.13898	0.48279
	1.24059	0.92714	0.73165	0.33295
SE	0.81793	0.6284	0.50475	0.23465
	0.95103	0.7116	0.56108	0.25104
ESE	1.22014	0.87805	0.66539	0.2715
	1.49735	1.09784	0.84898	0.35495
E	1.66241	1.26503	1.00313	0.45677
	1.68753	1.32254	1.07521	0.50464
ENE	1.56755	1.20846	0.96987	0.45849
	1.23217	0.95485	0.77234	0.36778
NE	0.89068	0.69631	0.56584	0.26817
	1.01625	0.77102	0.61514	0.28467
NNE	1.33786	0.94874	0.70976	0.28187
	1.65219	1.21551	0.9388	0.37848
N	1.84154	1.38928	1.10125	0.51894
	1.94569	1.49723	1.2126	0.56782
NNW	1.95195	1.45915	1.13466	0.50129
	1.66239	1.24727	0.98153	0.43762
NW	1.40878	1.07966	0.86264	0.38646
	2.02007	1.55066	1.24362	0.57016
WNW	2.59957	2.00674	1.61822	0.78892
	3.12082	2.36146	1.88189	0.8849
W	3.48215	2.63075	2.05867	0.85542

3. Estimate air concentrations at receptors located 575m from the source. Because the ISC model simulates a gaussian plume in any given direction, we could estimate air concentrations at 575 m from the source (i.e., the furthest edge of the 275m by 275m field) by plotting the concentration data for the 300m, 400m, 500m, and 750m(tank) or 1000m(land treatment unit) distances on a log-log graph. These graphs are shown in Figure 2-5 for the tank and 2-6 for the land treatment unit. For a given increment of distance off the plume centerline, each line on the graph represents the change in concentration with distance from the source. We used these graphs to predict the concentrations at 575m by drawing a vertical line at 575m that intersects with each of the plotted lines, then reading from the graph the concentrations at various increments of distance off the plume centerline, 575m from the source.

4. Estimate air concentrations at receptors located 137.5m off of the plume centerline. The concentration profiles for receptor points at a given distance (for example, 300m from the source) could be used to interpolate the concentration at a distance 137.5m off of the centerline of the plume in either direction, that is, the concentrations at the lateral edges of the pasture. Each of the profiles shown in Figures 2-7 and 2-8 represents the change in concentration moving away from the plume centerline at a given distance from the source. As shown in the figure, we can derive the concentration at 137.5m on either side of the plume centerline by drawing a line at 137.5m. The resulting concentration estimates at 300, 400, 500, and 575 m from the source were read directly from the graph.

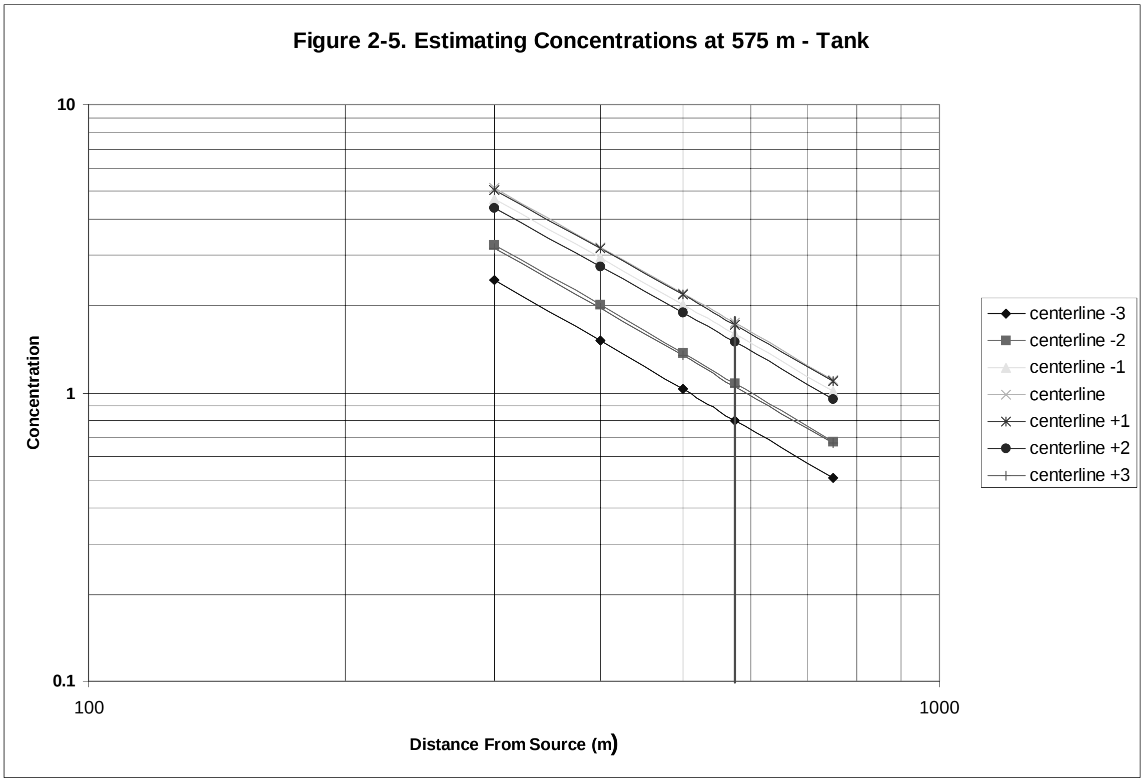


Figure 2-5. Estimating Concentrations at 575 m - Tank
10
1
0.1
Concentration
100
1000
Distance From Source (m)
centerline -3
centerline -2
centerline -1
centerline
centerline +1
centerline +2
centerline +3

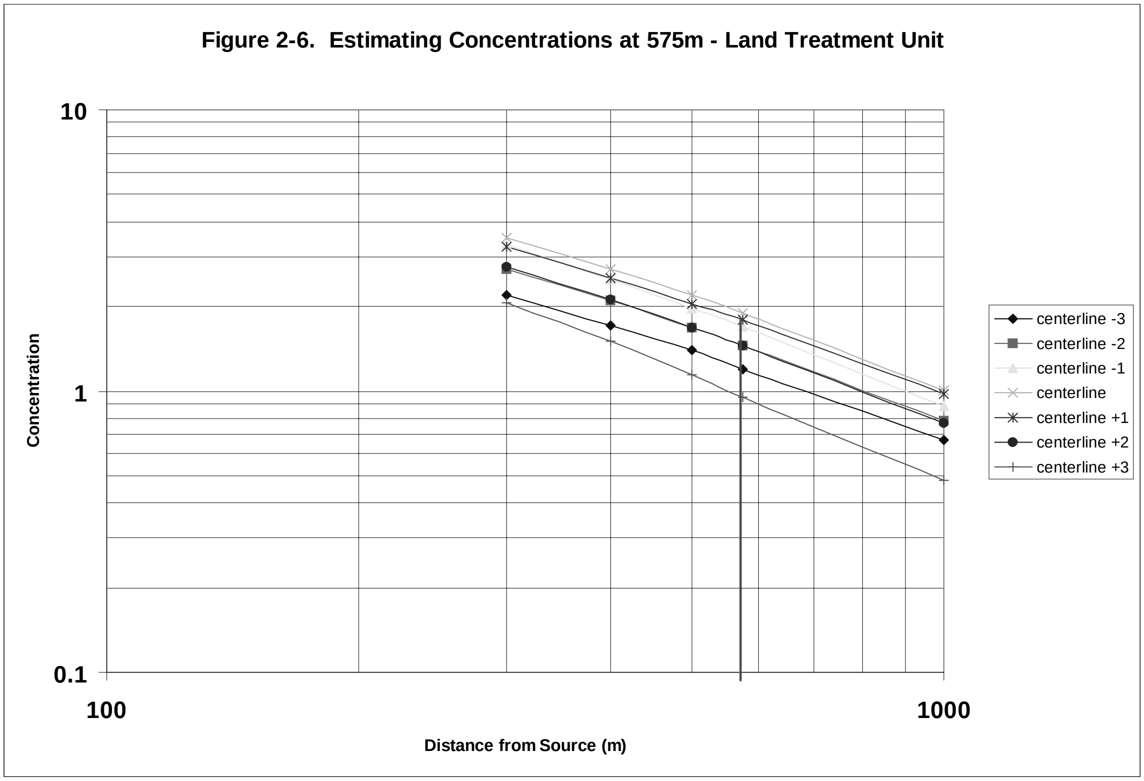


Figure 2-6. Estimating Concentrations at 575m - Land Treatment Unit
10
1
0.1
Concentration
100
1000
Distance from Source (m)
centerline -3
centerline -2
centerline -1
centerline
centerline +1
centerline +2
centerline +3

Figure 2-7. Estimating Concentrations at 137.5 m - Tank

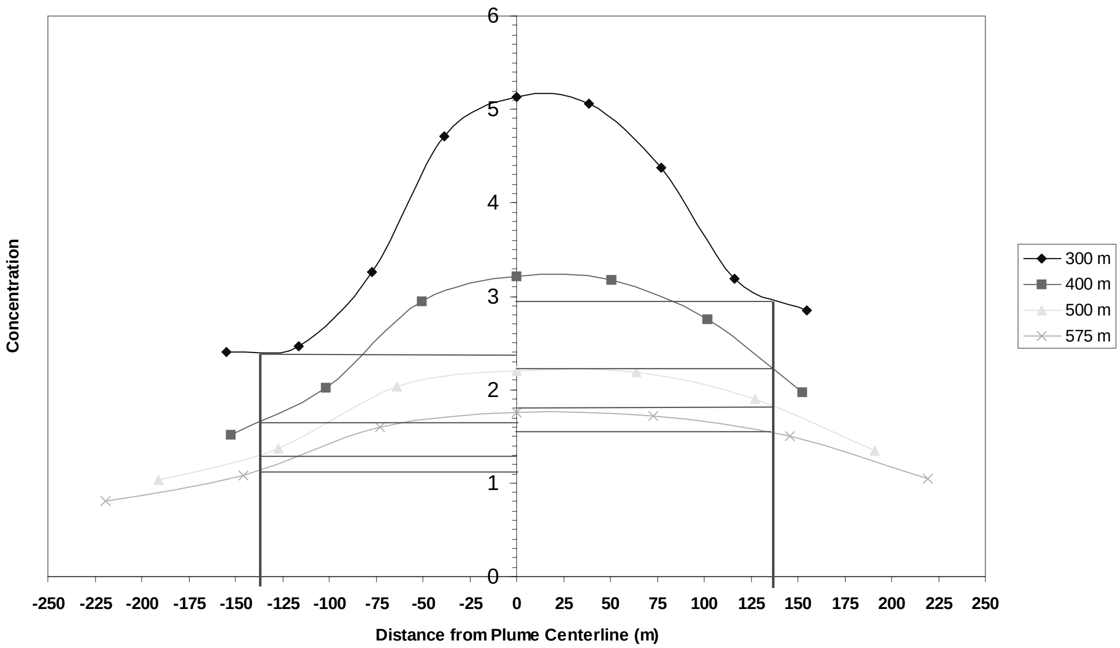
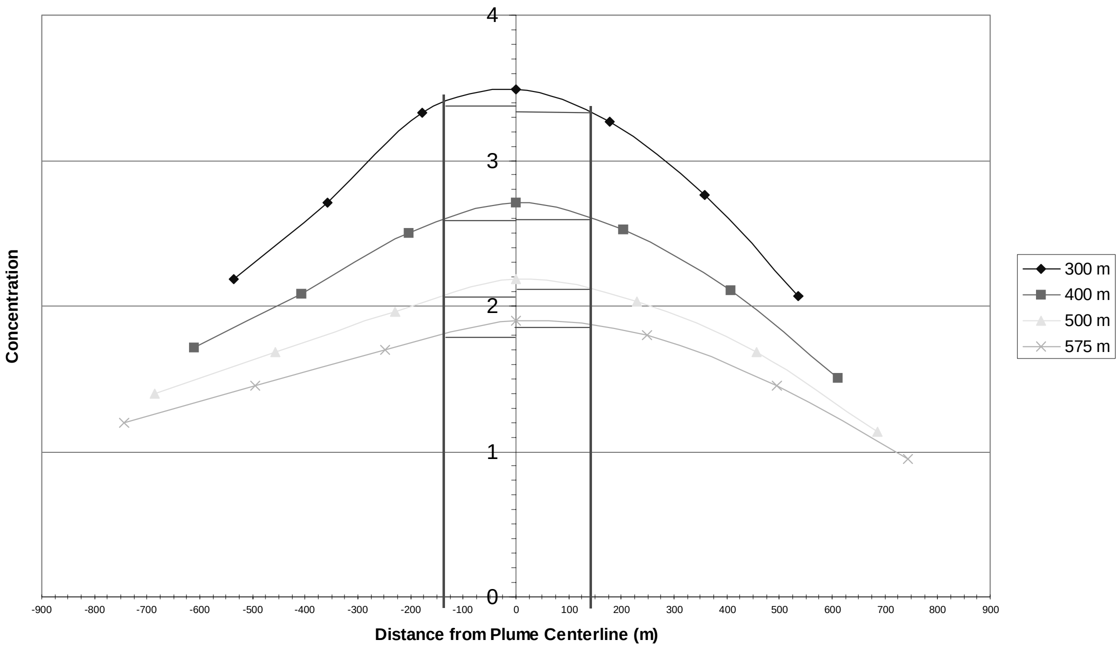


Figure 2-8. Estimating Concentrations at 137.5m - Land Treatment Unit



5. Calculate the average air concentration over a 275m by 275m pasture. Using the air concentrations derived in Steps 3 and 4 above, we created a matrix of air concentration points that represent the pasture, and performed the simple calculations needed to produce an area average. Figures 2-9 and 2-10 present the concentrations for the 275m by 275m pasture located 300m from a wastewater tank and the land treatment unit, respectively (for convenience, values were rounded at two decimal places). Tables 2-6 and 2-7 show the calculations. For each distance from the source (that is, 300, 400, 500, and 575m), only those air concentrations that fall within the 275m width were used to calculate the distance-specific average. For example, for the tank, we used nine receptor locations to estimate the average concentration at 300m from the source. The concentrations corresponding to adjacent points were averaged, and then the resulting concentrations were weighted according to the increment of distance they represent, to calculate an average concentration at 300 meters, as follows:

$$\frac{[(4.921+3.986+2.862+5.10+4.725+3.781)x38.7]+[(2.430+3.065)x21.4]}{275}$$

To avoid biasing the data by the number of receptor locations, we calculated the average concentration at each distance from the source prior to generating a pasture area average. For example, for the tank, the concentrations corresponding to adjacent distances from the source were averaged, and the resulting concentrations were weighted according to the increment of distance they represent, to calculate a pasture area average concentration, as follows:

$$\frac{[(3.363+2.355)x100]+(1.79x75)}{275}$$

For the tank, dividing the maximum concentration at 300m (5.13 ug/m^3 / mg/s-m^2, the concentration we used in the 1999 risk analysis) into the pasture area average of 2.568 ug/m^3 / mg/s-m^2 results in a ratio of 0.50 or a 50% reduction in the air concentration, therefore in risk.

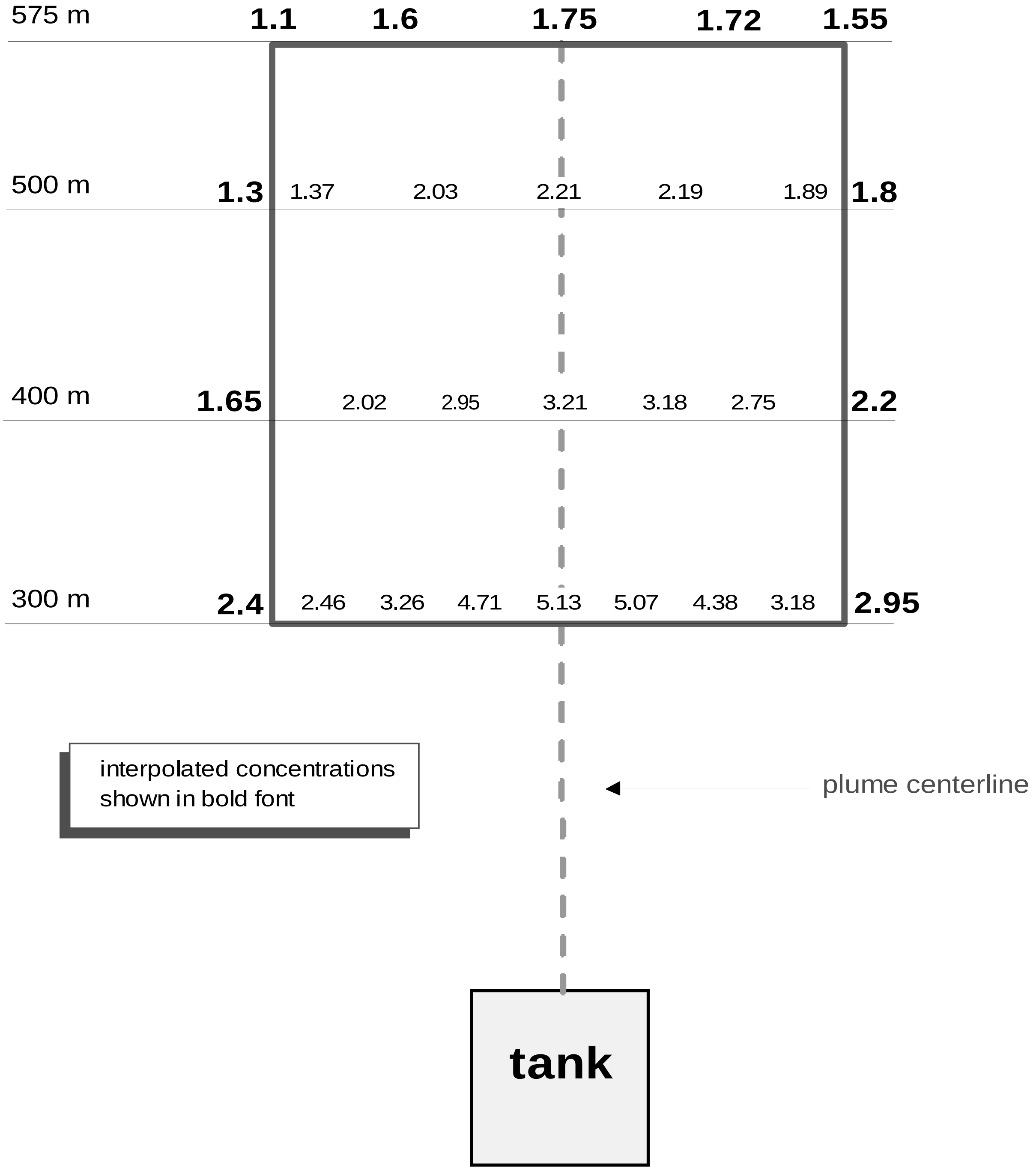


Figure 2-9. Air Concentrations for 275 by 275 m Pasture Located 300 m from Wastewater Tank (concentrations in ug/m3 / mg/s-m2)

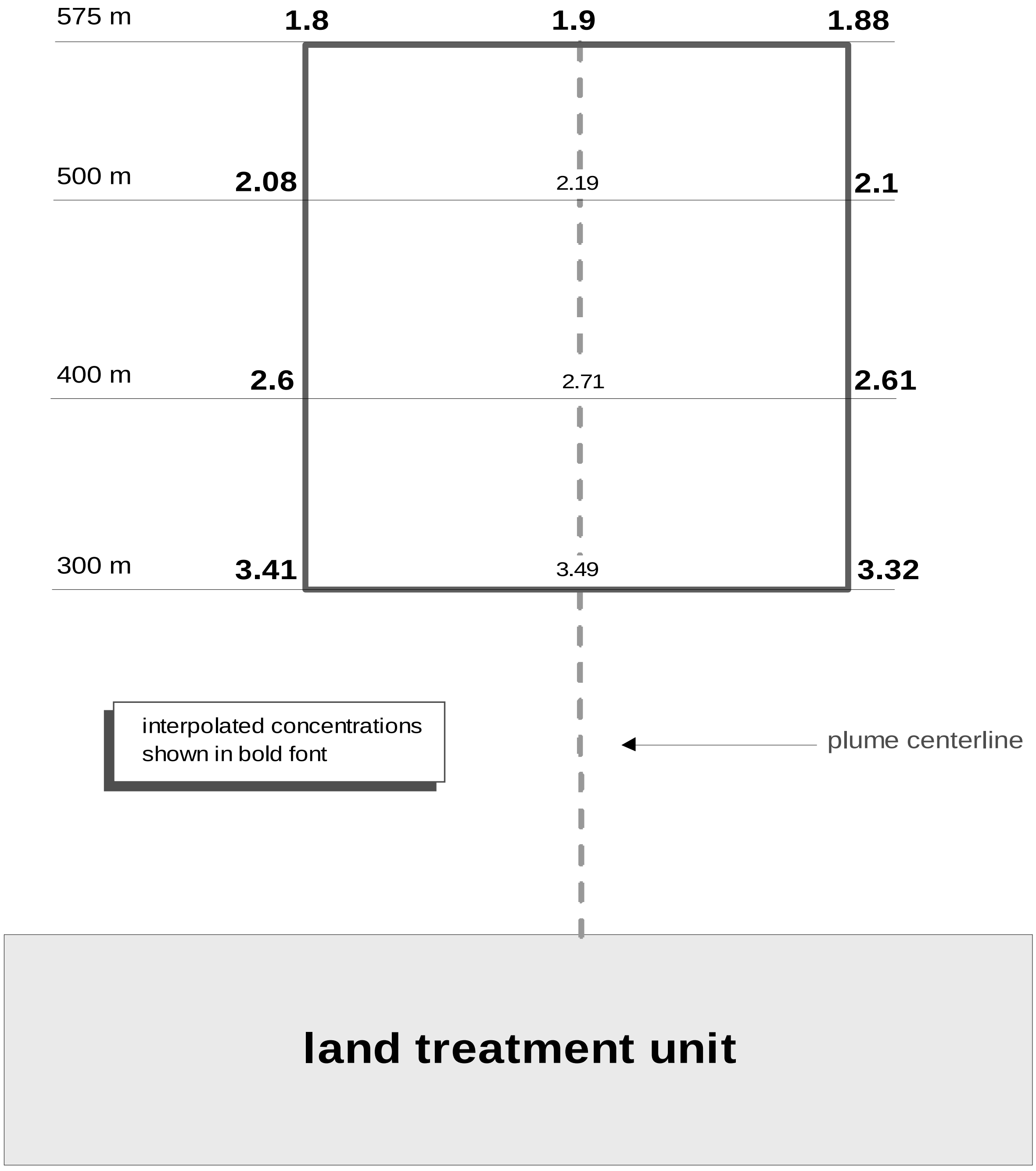


Figure 2-10. Air Concentrations for 275 by 275 m Pasture Located 300 m from Land Treatment Unit (concentrations in ug/m3 / mg/s-m2)

Table 2-6. Calculation of Average Concentration Over 275m by 275m Pasture – Tank

300 m			400 m			500 m			575 m		
Distance from Plume Centerline	Concentration ug/m^3/mg/s-m^2	Average Concentration Between Adjacent Points	Distance from Plume Centerline	Concentration ug/m^3/mg/s-m^2	Average Concentration Between Adjacent Points	Distance from Plume Centerline	Concentration ug/m^3/mg/s-m^2	Average Concentration Between Adjacent Points	Distance from Plume Centerline	Concentration ug/m^3/mg/s-m^2	Average Concentration Between Adjacent Points
-154.8	2.40257										
-137.5	**2.4**		-152.7	1.51459		-191.1	1.03091		-219	0.8	
		2.43007									
-116.1	2.46014		**-137.5**	**1.65**		**-137.5**	**1.3**		-146	1.08	
		2.86167			1.83275			1.33734			
-77.4	3.26319		-101.8	2.0155		-127.4	1.37468		**-137.5**	**1.1**	
		3.98659			2.48137			1.70131			1.35000
-38.7	4.70998		-50.9	2.94723		-63.7	2.02793		-73	1.6	
		4.92068			3.07802			2.11686			1.67500
0	5.13137		0	3.2088		0	2.20578		0	1.75	
		5.09966			3.19255			2.19644			1.73500
38.7	5.06794		50.9	3.1763		63.7	2.18709		73	1.72	
		4.72462			2.96224			2.04062			1.63500
77.4	4.3813		101.8	2.74817		127.4	1.89414		**137.5**	**1.55**	
		3.78104			**2.47409**			**1.84707**			
116.1	3.18078		**137.5**	**2.2**		**137.5**	**1.8**		146	1.5	
		3.06539									
137.5	**2.95**		152.7	1.96877		191.1	1.34505		219	1.05	
154.8	2.85208										
distance-weighted average concentration		3.998			2.727			1.983			1.605

distance-weighted pasture average	2.568
300m maximum ÷ pasture average:	0.50

Table 2-7. Calculation of Average Concentration Over 275m by 275m Pasture – Land Treatment Unit

300 m			400 m			500 m			575 m		
Distance from Plume Centerline	Concentration ug/m^3/mg/s-m^2	Average Concentration Between Adjacent Points	Distance from Plume Centerline	Concentration ug/m^3/mg/s-m^2	Average Concentration Between Adjacent Points	Distance from Plume Centerline	Concentration ug/m^3/mg/s-m^2	Average Concentration Between Adjacent Points	Distance from Plume Centerline	Concentration ug/m^3/mg/s-m^2	Average Concentration Between Adjacent Points
-178.7	3.33091		-203.7	2.50543		-228.7	1.96314		-248	1.7	
-137.5	**3.41**		**-137.5**	**2.6**		**-137.5**	**2.08**		**-137.5**	**1.8**	
		3.4518			2.6562			2.1326			1.8500
0	3.49363		0	2.71231		0	2.1852		0	1.9	
		3.4068			2.6612			2.1426			1.8900
137.5	**3.32**		**137.5**	**2.61**		**137.5**	**2.1**		**137.5**	**1.88**	
178.7	3.26376		203.7	2.52376		228.7	2.03361		248	1.8	
distance-weighted average concentration		3.429			2.659			2.138			1.87

distance-weighted pasture average:	2.525
300m maximum ÷ pasture average:	0.72

For the land treatment unit, dividing the maximum concentration at 300m (3.49 ug/m^3 / mg/s-m^2, the concentration we used in the 1999 risk analysis) into the pasture area average of 2.525 ug/m^3 / mg/s-m^2 results in a ratio of 0.72 or a 28% reduction in the air concentration, therefore in the portion of the risk estimate attributable to the air pathway (part of the nongroundwater pathway risk associated with the land treatment unit is due to the erosion/runoff pathway).

Section 5.3 summarizes how assuming a larger pasture size would influence our high end deterministic risk estimates for the aerated biological treatment tank and land treatment unit waste management scenarios.

2.3 Evaluation of the Exposure Scenario as it Relates to the Probabilistic Analyses

We stated on page 2-38 of the 1999 Risk Assessment TBD that in our probabilistic analyses we assumed that individuals (receptors) live between 50 and 1000 meters from the waste management unit, and randomly evaluated receptors located 50, 75, 100, 200, 300, 500, or 1000m from the waste management unit. In evaluating commenters' concerns regarding the overly conservative nature of our exposure assumptions, we acknowledged that the distances we chose to represent the location of receptors relative to the waste management units were too heavily weighted toward locations close to the waste management units. That is, the distances between 50m and 100m are 25 meters apart; the distances between 100m and 300m are 100 meters apart; the distance between 300m and 500m is 200 meters apart; and the distance between 500m and 1000m is 500 meters apart. As a result, our probabilistic risk percentiles likely are somewhat lower than they would have been if the distances we evaluated had been spaced equally. We expect this effect to be more pronounced for the wastewater tank scenario, since the air concentrations associated with the plume from the tank vary more greatly with distance than those for the land treatment unit. This is because the tank source is so much smaller than the land treatment unit source. Any overestimate of our probabilistic risk results is not particularly consequential since our listing decisions were based primarily on the results of the deterministic analyses, which, as discussed in this Addendum, we reevaluated in response to commenters' concerns.

3. ESTIMATING EXPOSURE POINT CONCENTRATIONS

This section provides technical information that supplements that provided in Section 3 of the 1999 Risk Assessment Technical Background Document. Section 3 of the background document described how EPA used models to estimate contaminant concentrations at a receptor's point of exposure.

The information we provide in this section includes:

- A discussion of how the removal of solids from wastewater prior to aerated biological treatment would influence the risk estimates for chlorinated aliphatics wastewaters; and
- A mass balance correction for the land treatment unit erosion pathway analysis.

3.1 Removal of Solids from Wastewater Prior to Aerated Biological Treatment

To evaluate the risks from management of wastewaters, EPA modeled the contaminant emissions from aerated biological wastewater treatment tanks. In the proposed rule EPA explained that significant risk was associated with exposure to dioxins emitted from aerated biological treatment tanks. Commenters on the proposed rule asserted that EPA failed to account for the fact that almost all of the dioxins in wastewaters are sorbed to solids and are removed in primary clarifiers prior to aeration. EPA agrees with the commenters concerns that we failed to accurately account for the fact that in aerated biological wastewater treatment systems, at least some solids removal generally will occur between the headworks of the wastewater treatment system and the influent to an aerated biological treatment tank. In the preamble to the proposed rule, EPA specifically stated that we selected wastewater data for evaluation that we believed represented the concentrations of contaminants in wastewaters at the influent (headworks) of treatment systems that are used to manage only wastewaters from the production of chlorinated aliphatic chemicals ("dedicated" chlorinated aliphatics wastewater samples; 64 FR 46483). In retrospect, our assumption that the same data that represent contaminant concentrations at the headworks of wastewater treatment systems could represent contaminant concentrations at the influent to aerated biological wastewater treatment tanks was somewhat flawed.

We reviewed information previously provided to us in industry survey responses and determined that of the eleven facilities that employ aerated biological processes to treat their wastewaters, nine employ primary clarification or other processes that have the effect of removing solids from wastewaters prior to their discharge to aerated biological treatment tanks. (One of these nine facilities is the facility from which we collected the "high end" wastewater sample used in the risk analysis that served as the basis for our proposed listing decision.) The remaining two facilities perform wastewater equalization in tanks prior to aerated biological treatment. One of these two facilities also employs wastewater pH adjustment with resultant precipitation of metal hydroxides prior to aerated biological treatment[2]. Both of these processes are expected to result in at least some solids removal from the wastestream. Moreover, EPA does not anticipate that treatment of the wastewaters in units such as primary clarifiers and equalization basins would result in dioxin air emissions greater than those that we originally predicted from aerated biological treatment tanks, because primary clarifiers are, by design, quiescent units (Metcalf and Eddy, 1991, p. 472), and we have no information that leads us to believe that the equalization tanks in use by the facilities are agitated.

To model the aerated biological treatment tanks correctly, that is, to determine what the appropriate influent concentration to the biological treatment tank should be, would have required that EPA model the wastewater treatment train from the point where wastewater enters the headworks of the treatment system to the point where the wastewater enters the aerated biological tank. Metcalf and Eddy (1991, p. 473) state that "efficiently designed and operated primary sedimentation tanks should remove from 50 to 70 percent of the suspended solids..." from wastewater. Consequently, we estimated how our emissions estimates would differ if we assumed alternately that 50 percent and 70 percent of the solids in chlorinated aliphatics wastewater analysis we used in our high end analysis were removed prior to discharge to the aerated biological treatment tank. Specifically, we assumed that the dioxins in

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[2] See the Listing Background Document for the Chlorinated Aliphatics Listing Determination (Final Rule), June 30, 2000, for a description of this information.

the solid and dissolved phases of the wastewater were in equilibrium, and that we could predict the concentration of dioxins sorbed to the solids through a linear equilibrium partitioning relationship. Assuming such a relationship, the total concentration of a congener in the wastewater sample as measured, C_{tot0}, is:

$$C_{tot_0} = C_{liq} - C_{solids}[TSS] = C_{liq}\ (1 - K_d[TSS]) \qquad \textbf{(3)}$$

Where

C_{tot0} is the initial total concentration of the congener in the wastewater

C_{liq} is the dissolved phase concentration of the congener in wastewater

C_{solids} is the solids phase concentration of the congener in wastewater

[TSS] is the initial suspended solids concentration of the congener in wastewater

K_d is the equilibrium partition coefficient for the congener, calculated using the following equation:

$$K_d = K_{oc}\ f_{oc} \qquad \textbf{(4)}$$

where

f_{oc}, the fraction of organic carbon, is assumed to be 0.03[3]

K_{oc}, the organic carbon partition coefficient, is calculated using the relationship of Karickhoff et al. (1979):

$$\log K_{oc} = -0.21 - \log K_{ow} \qquad \textbf{(5)}$$

where the values for K_{ow} are presented in Appendix C of the July 30, 1999 Risk Assessment Technical Background Document.

Therefore, the total influent congener concentration after 70% TSS removal, C_{tot1}, is:

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[3] In public comments, industry commenters conservatively assumed that the wastewater solids are only 3% organic carbon.

$$C_{tot_1} = C_{liq} \ (1 - 0.3 \ K_d \ [TSS]) \qquad \textbf{(6)}$$

If Equation 1 is written in terms of C_{liq}, then Equation 4 can be solved to determine the total congener concentration (dissolved phase plus solid phase) entering the aerated tank after 70% of the TSS is removed, as follows:

$$C_{tot_1} = \frac{C_{tot_0} \ (1 - 0.3 \ K_d \ [TSS])}{(1 - K_d \ [TSS])} \qquad \textbf{(7)}$$

Determining the total congener concentration entering the aerated tank after 50% of the TSS is removed requires substituting the value 0.5 for 0.3 in Equations 4 and 5. In Equation 5, the expression "$C_{tot0}/(1+K_d[TSS])$" represents the influent dissolved-phase congener concentration that is in equilibrium with the influent solids (TSS). For OCDF, this concentration exceeded the solubility limit. As a result, we used the OCDF solubility concentration in our analysis in lieu of the value calculated using the "$C_{tot0}/(1+K_d[TSS])$" expression.

After calculating C_{tot1} for each congener, we then computed, for each congener, the ratio of the uncorrected tank influent concentration we used in our original analysis, C_{tot0} (the analytical results for sample GL-02) and the tank influent concentration calculated assuming solids are removed prior to biological treatment, C_{tot1}. C_{tot0}/C_{tot1} is the factor by which the concentrations influent to the tank would differ from the value we used in our original analysis, that is, the ratio of the influent concentration presented in our 1999 Risk Assessment TBD to the influent concentration corrected for loss of dioxins due to removal of solids from the wastewater.

Although reducing the dioxin concentration influent to the tank by removing solids (and the sorbed dioxin phase) results in an overall decrease in the dioxin concentrations influent to the tank, therefore a decrease in dioxin emissions estimates, a smaller increase in emissions also occurs due to reducing the TSS concentration in the tank, as shown in Table D.3-3 of Appendix D.3 in the 1999 Risk Assessment Technical Background Document. This increase in emissions occurs because reducing the TSS influent to the tank reduces the mass of solids in the tank onto which dioxins can sorb. Therefore, the last step of the evaluation is to correct the emissions estimate to account for this emissions increase. We approximated this increase in emissions from the results of the sensitivity analyses for dioxins that we generated in developing Table D.3-3 of the 1999 Risk Assessment TBD. These sensitivity results are presented below in Table 3-1. Specifically, we assumed that removing 50 percent of the TSS results in increasing emissions by a factor of 1.25 and removing 70 percent of the TSS results in increasing emissions by a factor of 1.47. In the CHEMDAT8 model the relationship between TSS and emissions also is dependent on the solids balance with TOC. Consequently, the results of any sensitivity analysis will depend greatly on the characteristics of a given waste stream.

To account for both the decrease in emissions due to the reduced concentrations of dioxins entering the tank and the smaller increase in emissions due to the reduced solids concentration in the tank, we divided the value for C_{tot0}/C_{tot1} calculated above by 1.25 (for 50 percent solids removal) or 1.47 (for 70 percent solids removal) to obtain the values in Tables 3-2 and 3-3. The values presented in Tables 3-2 and 3-3 are our estimates of the factors by which our emissions estimates presented in the 1999 Risk Assessment TBD exceed the emissions predicted given the assumption that 70 percent and 50 percent of the solids, respectively, are removed from the wastewater prior to aerated biological treatment. Dividing our 1999 congener-specific risk estimates by the factors presented in Tables 3-2 and 3-3 produces revised risk estimates that account for solids removal prior to aerated biological treatment.

Table 3-1. Sensitivity Results for the Wastewater Tank Analysis

Contaminants	**Baseline Emissions** Baton Rouge Met Station, Central Tendency Tank, Average Concentration	**Emissions when TSS (or Biomass Solids) in Influent Are:**			
		Increased by a factor of 2		**Decreased by a factor of 2**	
	(Mg/yr)	**(Mg/yr)**	**% difference**	**(Mg/yr)**	**% difference**
Acetone	6.74E-04	6.74E-04	0.0	6.74E-04	0.0
Allyl Chloride	1.35E-03	1.35E-03	0.0	1.35E-03	0.0
Aminobiphenyl,4-	3.68E-07	3.67E-07	-0.1	3.68E-07	0.1
Benzoic Acid	2.24E-05	2.24E-05	-0.1	2.24E-05	0.0
Benzyl Alcohol	7.09E-07	7.08E-07	-0.1	7.09E-07	0.0
Bis(2-chloroethyl)ether	6.18E-04	6.18E-04	-0.1	6.19E-04	0.0
Bis(2-chloroisopropyl)ether	3.20E-04	3.20E-04	-0.1	3.20E-04	0.0
Bis(2-ethylhexyl)phthalate	5.15E-08	3.51E-08	-31.9	6.73E-08	30.6
Bromodichloromethane	4.24E-04	4.24E-04	0.0	4.24E-04	0.0
Bromoform	1.47E-04	1.47E-04	-0.1	1.47E-04	0.0
Carbon Disulfide	9.35E-04	9.35E-04	0.0	9.35E-04	0.0
Chlorobenzene	2.14E-04	2.14E-04	0.0	2.14E-04	0.0
Chloro-1,3,-butadiene,2-	1.97E-03	1.97E-03	0.0	1.97E-03	0.0
Chloroethane (Ethyl Chloride)	1.79E-03	1.79E-03	0.0	1.79E-03	0.0
Chloroform	4.28E-02	4.28E-02	0.0	4.28E-02	0.0
Cresol(-o)	1.49E-07	1.49E-07	0.0	1.49E-07	0.0
Cresol(-p)	1.14E-07	1.14E-07	0.0	1.14E-07	0.0
Dibromochloromethane	6.57E-04	6.56E-04	-0.2	6.58E-04	0.1
Dichloroethane(1,2)	3.98E-03	3.98E-03	0.0	3.98E-03	0.0
Dichloroethene 1,2 Trans	6.63E-04	6.62E-04	0.0	6.63E-04	0.0
Dichloroethylene, 1,2 - Cis	8.10E-04	8.10E-04	0.0	8.10E-04	0.0
Diethyl Phthalate	2.10E-06	2.10E-06	-0.1	2.10E-06	0.0
Dimethyl Phthalate	1.90E-08	1.90E-08	0.0	1.90E-08	0.0
Di-n-octyl Phthalate	4.87E-06	3.20E-06	-34.3	6.58E-06	35.2
Ethylbenzene	3.64E-04	3.64E-04	-0.1	3.64E-04	0.0
Hexachlorobenzene	1.17E-03	1.03E-03	-11.4	1.25E-03	6.9
HpCDD, 1,2,3,4,6,7,8-	2.31E-10	1.46E-10	-36.6	3.24E-10	40.7
HpCDF, 1,2,3,4,6,7,8-	3.93E-08	2.50E-08	-36.4	5.48E-08	39.4
HpCDF, 1,2,3,4,7,8,9-	8.45E-09	5.39E-09	-36.2	1.18E-08	39.7
HxCDD, 1,2,3,4,7,8-	5.33E-11	3.39E-11	-36.4	7.47E-11	40.2
HxCDD, 1,2,3,6,7,8-	1.35E-10	8.62E-11	-36.0	1.87E-10	39.1
HxCDD, 1,2,3,7,8,9-	1.38E-10	8.81E-11	-36.0	1.91E-10	39.1
HxCDF, 1,2,3,4,7,8-	7.15E-09	4.58E-09	-35.9	9.94E-09	39.0
HxCDF, 1,2,3,6,7,8-	1.65E-09	1.06E-09	-36.1	2.30E-09	39.3
HxCDF, 1,2,3,7,8,9-	9.06E-10	5.80E-10	-36.0	1.26E-09	39.2
HxCDF, 2,3,4,6,7,8-	1.54E-09	9.87E-10	-36.0	2.15E-09	39.2
Mercury	1.14E-04	1.13E-04	-1.2	1.15E-04	1.0
Methyl Ethyl Ketone, 2-Butanone	6.25E-04	6.24E-04	-0.2	6.25E-04	0.1
Methylene Chloride, Dichloromethane	1.26E-03	1.26E-03	0.0	1.26E-03	0.0
OCDD, 1,2,3,4,5,7,8,9-	5.18E-12	3.29E-12	-36.4	7.26E-12	40.2
OCDF, 1,2,3,4,6,7,8,9-	1.11E-09	7.02E-10	-36.8	1.57E-09	41.0
PeCDF, 2,3,4,7,8-	6.55E-10	4.23E-10	-35.4	9.03E-10	37.8
Pentachlorophenol	4.39E-08	4.36E-08	-0.8	4.41E-08	0.4
Phenol	4.98E-07	4.98E-07	0.0	4.98E-07	0.0
Styrene	1.14E-03	1.14E-03	-0.1	1.14E-03	0.1
TCDD, 2,3,7,8-	2.48E-11	1.62E-11	-34.6	3.37E-11	36.0
TCDF, 2,3,7,8-	1.48E-10	9.75E-11	-34.2	2.00E-10	35.1
Tetrachloroethene	8.29E-04	8.29E-04	0.0	8.29E-04	0.0
Trichloroethylene	8.36E-06	8.36E-06	0.0	8.36E-06	0.0
Trichlorophenol 2,4,5	1.63E-06	1.63E-06	-0.1	1.63E-06	0.1
Trichlorophenol 2,4,6	1.46E-04	1.44E-04	-1.3	1.47E-04	0.7
Average impact as compared to Baseline emissions		**12% decrease**		**13% increase**	

Table 3-2. Ratio of the 1999 High End Emissions Estimate and Emissions Estimate that Assumes Solids (TSS) Removal Prior to Aerated Biological Treatment 70% TSS Removal Efficiency	
1,2,3,4,6,7,8-HpCDD	2.26
1,2,3,4,6,7,8-HpCDF	0.07
1,2,3,4,7,8,9-HpCDF	0.26
1,2,3,4,7,8-HxCDD	2.25
1,2,3,6,7,8-HxCDD	2.22
1,2,3,7,8,9-HxCDD	2.25
1,2,3,4,7,8-HxCDF	2.22
1,2,3,6,7,8-HxCDF	2.22
1,2,3,7,8,9-HxCDF	0
2,3,4,6,7,8-HxCDF	2.22
2,3,4,7,8-PeCDF	2.16
2,3,7,8-TCDD	2.08
2,3,7,8-TCDF	2.03
OCDD	0.024
OCDF	0.00063

Table 3-3. Ratio of the 1999 High End Emissions Estimate and Emissions Estimate that Assumes Solids (TSS) Removal Prior to Aerated Biological Treatment 50% TSS Removal Efficiency	
1,2,3,4,6,7,8-HpCDD	1.60
1,2,3,4,6,7,8-HpCDF	0.05
1,2,3,4,7,8,9-HpCDF	0.19
1,2,3,4,7,8-HxCDD	1.60
1,2,3,6,7,8-HxCDD	1.59
1,2,3,7,8,9-HxCDD	1.60
1,2,3,4,7,8-HxCDF	1.59
1,2,3,6,7,8-HxCDF	1.59
1,2,3,7,8,9-HxCDF	0
2,3,4,6,7,8-HxCDF	1.59
2,3,4,7,8-PeCDF	1.57
2,3,7,8-TCDD	1.54
2,3,7,8-TCDF	1.53
OCDD	0.017
OCDF	0.00045

Based on our calculations, removing solids from chlorinated aliphatics wastewaters prior to biological treatment might reduce the high end deterministic risk estimate by a factor of ranging from approximately 0.67 (for 70 percent removal of solids) to 0.94 (for 50 percent removal of solids). As explained on p. 3-2 of the 1999 Risk Assessment Technical Background Document, we originally constrained ("capped") the influent concentrations of congeners in the wastewaters at their aqueous solubility concentrations to account for the fact that dioxins are strongly hydrophobic and are expected to be sorbed to solids preferentially in the wastewater influent, thus are unlikely to exist in the dissolved phase in excess of their solubility limits (see also the Errata for Section 3 in this Addendum). Because the wastewaters contained both dissolved- and particle-phase dioxins (the samples were not filtered prior to analysis), we corrected our application of this constraint by applying it directly to the dissolved-phase congener concentrations (that is, the expression "$C_{tot0}/(1+K_d[TSS])$" from Equation 7). Consequently, the ratios presented in Tables 3-2 and 3-3 are less than 1 for 1,2,3,4,6,7,8-HpCDF, 1,2,3,4,7,8,9-HpCDF, OCDD, and OCDF. That is, in our revised analysis, the concentrations of these congeners in the wastewater exceeded their solubility limits. Therefore, the concentrations of these four congeners in the influent wastewater were greater in our reevaluation than they were in the 1999 analysis, although the concentrations of all of the other congeners decreased.

Section 5.3 summarizes how accounting for solids removal prior to aerated biological treatment of wastewaters would influence our high end deterministic risk estimates for chlorinated aliphatics wastewaters.

3.2 Mass Balance Correction for the Land Treatment Unit Erosion Pathway Analysis

To predict the contaminant concentration in a receiving field located near the land treatment unit, we developed a conceptual model for overland transport based on the assumption that the system (i.e., the land treatment unit, the field, and the buffer areas that comprise the drainage subbasin) is at steady state. The steady-state assumption was crucial to developing a modeling approach that could be implemented within a concatenation of spreadsheet models that are designed to model steady state processes. The overland transport model relies on the Universal Soil Loss Equation (USLE) to predict soil loads to the waterbody and receiving field; application of the USLE in this context implies a steady state system. Moreover, the use of a sediment delivery ratio to predict the fraction of eroded soil that is deposited in the area between the land treatment unit and the waterbody implicitly assumes that the sediment delivery does not change with time.

The overland transport model used the maximum 9-year average soil concentration predicted by the source partitioning model as the contaminant concentration in the land treatment unit. Because the system was assumed to be at steady state, we fixed the land treatment unit concentration during the period of the simulation, effectively creating an infinite source (defined as a source with a fixed concentration that does not deplete over time). However, the drainage subbasin is not really a steady state system. In reality, the land treatment unit concentration would decrease with time until the land treatment unit was "clean", and some or all of the contaminant on the receiving field would be transported to the stream via soil erosion and runoff mechanisms. This type of contaminant profile would not resemble steady state, but rather, a transient system with a concentration profile for the field that increases with time, reaches a peak, and then decreases with time. In other project analyses assuming an infinite source, we found that it takes roughly 50 years for the model to reach steady state but, as our calculations have demonstrated, the contaminant mass is exhausted well before that time for the land treatment unit. In essence our calculations show that the total land treatment unit emissions from soil erosion over an 80 year time period are roughly a factor of 9 above the emissions possible given the mass loaded over the active life of the unit. Table 3-4 provides this calculation. (Both the central tendency and high end emissions estimates for the land treatment unit are provided in detail in Tables 3-5 and 3-6.) This result is not unexpected because the source is not allowed to deplete over the simulation period so that steady state conditions may be approximated. Section 5.3 summarizes how including this factor in our exposure analyses would influence our high end deterministic estimate of risk.

The Agency recognizes this as a limitation of the overland transport model and has since developed a more rigorous modeling approach that simulates the changes and source and field concentration with time (that is, the model develops concentration profiles). However,

Table 3-4. Erosion Mass Balance Calculation – Land Treatment Unit (LTU)

Total High End Dioxin Load to LTU Available to be Emitted (g TCDD TEQ), based on 907 ng/kg waste concentration and 624 Mton/year waste generation rate:		
	Annual	**Over 40-year Active Life**
	0.57	23
High End Emissions (g TCDD TEQ) (from Table 3-6):		
	Annual	**Over 80-year Active and Inactive Life**
Vapor	0.107	8.56
Particulate (PM10)	0.0091	0.72
Particulate (PM30)	0.014	1.1
Erosion	1.4	109.6
Runoff	0.00027	0.022
Leaching	0.00005	0.004
Total	1.5	120.0
Mass TCDD (g TCDD TEQ) Actually Available for Erosion = Total Load to LTU - Vapor and Particulate Emissions - Runoff Emissions - Leaching Emissions:		
	Annual	**Over 80-year Active and Inactive Life**
	0.436	12.244
Factor by which erosion may be overestimated (TEQ erosion emissions/TEQ Available for Erosion): 9		

Table 3-5. Central Tendency TCDD TEQ Emission Estimates for the EDC/VCM Land Treatment Unit

Congener	TEF	Vapor emissions (g/m2-yr)	Vapor emissions (g/yr)	Particulate emissions (PM 10) (g/m2-yr)	Particulate emissions (PM 10) (g/yr)	Particulate emissions (PM 30) (g/m2-yr)	Particulate emissions (PM 30) (g/yr)	Erosion emissions (g/m2-yr)	Erosion emissions (g/yr)	Runoff emissions (g/m2-yr)	Runoff emissions (g/yr)	Leaching Emission (g/yr)	Total emissions (g/yr)
2,3,7,8-TCDD	1	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
1,2,3,4,5,7,8,9-OCDD	0.001	2.4E-11	1.7E-05	3.7E-11	2.6E-05	5.6E-11	3.8E-05	5.6E-09	3.9E-03	7.7E-13	5.3E-07	9.5E-8	3.9E-03
1,2,3,7,8,9-HxCDD	0.1	1.3E-10	8.9E-05	7.9E-12	5.5E-06	1.2E-11	8.2E-06	1.2E-09	8.2E-04	3.3E-13	2.3E-07	4.2E-8	9.3E-04
1,2,3,4,6,7,8-HpCDD	0.01	1.7E-10	1.2E-04	3.8E-11	2.6E-05	5.6E-11	3.9E-05	5.7E-09	3.9E-03	1.9E-13	1.3E-07	2.4E-8	4.1E-03
OCDF	0.001	2.1E-10	1.4E-04	1.8E-10	1.2E-04	2.7E-10	1.9E-04	2.7E-08	1.9E-02	2.3E-13	1.6E-07	2.8E-8	1.9E-02
1,2,3,4,7,8-HxCDD	0.1	1.2E-10	8.2E-05	1.3E-11	8.8E-06	1.9E-11	1.3E-05	1.9E-09	1.3E-03	1.7E-13	1.2E-07	2.1E-8	1.4E-03
1,2,3,7,8-PeCDD	0.5	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2,3,7,8-TCDF	0.1	4.6E-11	3.1E-05	1.3E-12	8.9E-07	1.9E-12	1.3E-06	2.0E-10	1.3E-04	3.1E-13	2.1E-07	4.0E-8	1.7E-04
1,2,3,4,7,8,9-HpCDF	0.01	9.8E-10	6.7E-04	5.8E-11	4.0E-05	8.7E-11	6.0E-05	8.8E-09	6.1E-03	5.9E-13	4.1E-07	7.5E-8	6.8E-03
2,3,4,7,8-PeCDF	0.5	1.4E-09	9.6E-04	7.5E-11	5.1E-05	1.1E-10	7.7E-05	1.1E-08	7.8E-03	7.3E-12	5.0E-06	9.3E-7	8.9E-03
1,2,3,7,8-PeCDF	0.05	1.2E-10	8.3E-05	5.6E-12	3.9E-06	8.5E-12	5.8E-06	8.5E-10	5.9E-04	7.4E-13	5.1E-07	9.5E-8	6.8E-04
1,2,3,6,7,8-HxCDF	0.1	1.5E-09	1.0E-03	1.2E-10	8.6E-05	1.9E-10	1.3E-04	1.9E-08	1.3E-02	5.2E-12	3.6E-06	6.5E-7	1.4E-02
1,2,3,6,7,8-HxCDD	0.1	1.8E-10	1.2E-04	1.1E-11	7.6E-06	1.7E-11	1.1E-05	1.7E-09	1.1E-03	4.6E-13	3.2E-07	5.8E-8	1.3E-03
2,3,4,6,7,8-HxCDF	0.1	1.6E-09	1.1E-03	1.0E-10	7.1E-05	1.5E-10	1.1E-04	1.6E-08	1.1E-02	4.3E-12	3.0E-06	5.4E-7	1.2E-02
1,2,3,4,6,7,8-HpCDF	0.01	5.0E-09	3.4E-03	3.0E-10	2.0E-04	4.4E-10	3.1E-04	4.5E-08	3.1E-02	3.0E-12	2.1E-06	3.8E-7	3.5E-02
1,2,3,4,7,8-HxCDF	0.1	2.7E-09	1.8E-03	1.5E-10	1.0E-04	2.3E-10	1.6E-04	2.3E-08	1.6E-02	6.3E-12	4.3E-06	7.9E-7	1.8E-02
1,2,3,7,8,9-HxCDF	0.1	8.5E-10	5.8E-04	5.6E-11	3.8E-05	8.4E-11	5.8E-05	8.4E-09	5.8E-03	2.3E-12	1.6E-06	2.9E-7	6.5E-03
Total TEQ		**1.5E-08**	**1.0E-02**	**1.2E-09**	**7.9E-04**	**1.7E-09**	**1.2E-03**	**1.7E-07**	**1.2E-01**	**3.2E-11**	**2.2E-05**	**4.1E-6**	**1.3E-01**

Table 3-6. High End TCDD TEQ Emission Estimates for the EDC/VCM Land Treatment Unit

Congener	TEF	Vapor emissions (g/m2-yr)	Vapor emissions (g/yr)	Particulate emissions (PM 10) (g/m2-yr)	Particulate emissions (PM 10) (g/yr)	Particulate emissions (PM 30) (g/m2-yr)	Particulate emissions (PM 30) (g/yr)	Erosion emissions (g/m2-yr)	Erosion emissions (g/yr)	Runoff emissions (g/m2-yr)	Runoff emissions (g/yr)	Leaching Emission (g/yr)	Total emissions (g/yr)
2,3,7,8-TCDD	1	1.4E-08	9.8E-03	4.3E-10	3.0E-04	6.4E-10	4.4E-04	6.5E-08	4.5E-02	7.9E-11	5.5E-05	9.6E-6	5.5E-02
1,2,3,4,5,7,8,9-OCDD	0.001	6.7E-11	4.6E-05	1.1E-10	7.4E-05	1.6E-10	1.1E-04	1.6E-08	1.1E-02	2.3E-12	1.5E-06	2.5E-7	1.1E-02
1,2,3,7,8,9-HxCDD	0.1	1.3E-09	8.7E-04	8.8E-11	6.0E-05	1.3E-10	9.1E-05	1.3E-08	9.1E-03	3.7E-12	2.5E-06	4.2E-7	1.0E-02
1,2,3,4,6,7,8-HpCDD	0.01	5.3E-10	3.7E-04	1.2E-10	8.6E-05	1.9E-10	1.3E-04	1.9E-08	1.3E-02	6.4E-13	4.4E-07	7.2E-8	1.4E-02
OCDF	0.001	3.9E-09	2.7E-03	3.5E-09	2.4E-03	5.3E-09	3.6E-03	5.3E-07	3.7E-01	4.5E-12	3.1E-06	5.1E-7	3.8E-01
1,2,3,4,7,8-HxCDD	0.1	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
1,2,3,7,8-PeCDD	0.5	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2,3,7,8-TCDF	0.1	4.6E-09	3.2E-03	1.7E-10	1.2E-04	2.6E-10	1.8E-04	2.6E-08	1.8E-02	4.0E-11	2.8E-05	4.8E-6	2.1E-02
1,2,3,4,7,8,9-HpCDF	0.01	2.8E-08	1.9E-02	1.9E-09	1.3E-03	2.9E-09	2.0E-03	2.9E-07	2.0E-01	1.9E-11	1.3E-05	2.2E-6	2.2E-01
2,3,4,7,8-PeCDF	0.5	1.4E-08	9.8E-03	8.8E-10	6.0E-04	1.3E-09	9.1E-04	1.3E-07	9.1E-02	8.6E-11	5.9E-05	1.0E-5	1.0E-01
1,2,3,7,8-PeCDF	0.05	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
1,2,3,6,7,8-HxCDF	0.1	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
1,2,3,6,7,8-HxCDD	0.1	1.7E-09	1.2E-03	1.2E-10	8.1E-05	1.8E-10	1.2E-04	1.8E-08	1.2E-02	4.9E-12	3.4E-06	5.6E-7	1.4E-02
2,3,4,6,7,8-HxCDF	0.1	1.2E-08	8.6E-03	9.3E-10	6.4E-04	1.4E-09	9.6E-04	1.4E-07	9.7E-02	3.9E-11	2.7E-05	4.5E-6	1.1E-01
1,2,3,4,6,7,8-HpCDF	0.01	4.3E-08	3.0E-02	2.9E-09	2.0E-03	4.4E-09	3.0E-03	4.4E-07	3.0E-01	3.0E-11	2.0E-05	3.4E-6	3.4E-01
1,2,3,4,7,8-HxCDF	0.1	3.1E-08	2.1E-02	2.0E-09	1.4E-03	3.0E-09	2.1E-03	3.0E-07	2.1E-01	8.3E-11	5.7E-05	9.6E-6	2.3E-01
1,2,3,7,8,9-HxCDF	0.1	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Total TEQ		**1.6E-07**	**1.1E-01**	**1.3E-08**	**9.1E-03**	**2.0E-08**	**1.4E-02**	**2.0E-06**	**1.4E+00**	**3.9E-10**	**2.7E-04**	**4.6E-5**	**1.5E+00**

High end parameters: Contaminant concentration and exposure duration.

for the purposes of this risk assessment, the Agency considers the overland transport model to provide a reasonable (although protective) estimate of the concentration in the field (and waterbody) for a 9-year exposure duration. The movement of contaminant through the system is likely to produce a concentration gradient that moves with time from the land treatment unit, through the buffer area, to the receiving field, and eventually to the waterbody. Because the emission estimates from the land treatment unit indicate that the contaminant mass is not depleted during a 9-year exposure duration, the steady-state calculation of the field concentration using a 9-year average concentration is appropriate. The output concentration for the receiving field calculated by the overland transport model is used directly to predict risks from direct ingestion of soil and through the consumption of contaminated food grown/raised on the field. These calculations are not time dependent and, therefore, fixing the concentration at the maximum 9-year average in the land treatment unit is an appropriate way to predict risks for this scenario. Over a 30-year exposure duration, it is likely that this approach slightly overpredicts the risks (using the maximum 30-year average concentration for the land treatment unit). Over a 30-year duration, the mass balance of total emissions would be violated (that is, more mass would leave the land treatment unit than managed in the land treatment unit). However, it is important to remember that the system is assumed to be at steady state and that the field concentration is a function of the starting concentration in the land treatment unit which is fixed through time. As a result, the effective exposure duration would be reduced from 30 years to 15 years when the contaminant mass is depleted (assuming the total mass managed in the land treatment unit during its active life is available during those 15 years).

4. EXPOSURE AND TOXICITY ASSESSMENTS

This section provides technical information that supplements that provided in Section 4 of the 1999 Risk Assessment TBD. Section 4 of the background document described how we estimated the magnitude, frequency, duration, and routes of exposure (the exposure assessment) and how we characterized the toxicity of contaminants of potential concern.

The information we provide in this section includes:

- A discussion concerning how beef ingestion rates should be corrected for beef cooking and post-cooking losses; and
- A summary of chloroform exposure point concentrations and a reevaluation of our chloroform toxicity assessment.

4.1 Cooking and Post-Cooking Loss of Beef

The intake rates that we used for the adult farmer (and certain child of farmer age cohorts) in the 1999 Risk Assessment TBD should have incorporated loss of beef due to cooking and post-cooking processes. The Exposure Factors Handbook (USEPA, 1997; “the Handbook”) explains that the intake rates it provides for home-produced food items in Chapter 13 do not reflect actual food consumption (intake), but instead were derived from the amount of household food consumption in an economic sense, that is, they are the measure of the weight of food brought into the household that has been consumed (used up) in some manner. The Handbook explains that in addition to food being consumed by individuals, food may be used up by spoiling, by being discarded (for example, inedible parts), through cooking processes, etc. The Handbook provides estimated preparation losses for beef that include cooking losses (which include dripping and volatile losses) and post-cooking losses (which include cutting, bones, excess fat, scraps, and juices.) The authors of the Handbook averaged these losses across all cuts and cooking methods to obtain a mean net cooking loss and a mean net post-cooking loss for beef. The Handbook explains that the preparation loss factors presented “are intended to convert intake rates based on ‘household consumption’ to rates reflective of what individuals actually consume. However, these factors do not include losses to spoilage, feeding to pets, food thrown away, etc."

For beef, the Handbook presents a mean net cooking loss of 27 percent and a mean net post-cooking loss of 24 percent (USEPA, 1997, Table 13-5). As explained in the Handbook, the intake rates tabulated in Chapter 13 of the Handbook can be adjusted to reflect actual consumption using the following equation:

$$I_A = I x (1 - L_1) x (1 - L_2)$$

where:

I_A is the adjusted intake rate
I is the tabulated intake rate
L_1 is the cooking loss
L_2 is the post-cooking loss.

Therefore, for beef, the adjusted intake rate is a factor of 0.55 times the tabulated intake rate. Because the risk equation is "linear" (see USEPA, 1999, Appendix E, Table E-5.8) any adjustment of intake rate correlates directly to an adjustment in the risk estimate. Therefore, for our adult farmer scenario (and certain age cohorts of our child of farmer scenario), the estimate of risk attributable to ingestion of beef should be modified by a factor of 0.55 (because this value modifies risk from beef ingestion only, the factor modifying the total risk estimate would be greater).

Section 5.3 summarizes how accounting for cooking and post-cooking loss of beef would influence our high end deterministic risk estimates for chlorinated aliphatic wastewaters and EDC/VCM sludges managed in land treatment unit.

4.2 Chloroform Exposure Point Concentrations and Toxicity Assessment

The 1999 Risk Assessment TBD and Addendum provided results of an assessment of cancer risk due to inhalation of chloroform emitted from wastewaters. The cancer slope factor used in this analysis was based extrapolating response data in the low dose range using a linear approach called the linearized multistage (LMS) model. However, based on an evaluation initiated by EPA's Office of Water (OW), we now believe the weight of evidence for the carcinogenic mode of action for chloroform does not support a mutagenic mode of action, therefore a nonlinear low dose extrapolation is more appropriate for assessing risk from exposure to chloroform. EPA's Science Advisory Board (SAB), the World Health Organization (WHO), and the Society of Toxicology all strongly endorse a nonlinear approach for assessing risk from chloroform.

Although OW conducted their evaluation of chloroform carcinogenicity for oral exposure, the nonlinear approach for low-dose extrapolation would apply to inhalation exposure to chloroform as well, since chloroform's mode of action is understood to be the same for both ingestion and inhalation exposures. Specifically, tumorgenesis for both ingestion and inhalation exposures is induced through cytotoxicity (cell death) produced by the oxidative generation of highly reactive metabolites (phosgene and hydrochloric acid), followed by regenerative cell proliferation (63 *FR* 15685). Chloroform-induced liver tumors in mice are only seen after bolus, corn oil dosing, and are not found following administration by other routes (drinking water and inhalation). Rat liver tumors are not induced by chloroform following either drinking water or corn oil gavage administration. Kidney tumors are found in mice exposed to chloroform via inhalation or in toothpaste preparations, and in rats when exposed via drinking water or corn oil gavage. However, kidney and liver tumors develop only at doses that cause persistent cytotoxicity and regenerative proliferation, regardless of route of exposure or dosing regime. The overall dose-response for the cytotoxicity and cell proliferation responses is nonlinear (ILSI, 1997, Templin et al., 1996a,b,c; and Templin et al.,1998).

As explained in EPA OW's March 31, 1998, and December 16, 1998, Federal Register notices pertaining to chloroform (63 *FR* 15673 and 63 *FR* 69389, respectively), EPA now believes that "based on the current evidence for the mode of action by which chloroform may cause tumorgenesis,...a nonlinear approach is more appropriate for extrapolating low dose cancer risk rather than the low dose linear approach..."(63 *FR* 15685). In fact, OW determined that given chloroform's mode of carcinogenic action, liver toxicity (a noncancer health effect) actually “is a more sensitive effect of chloroform than the induction of tumors” and that protecting against liver toxicity “should be protective against carcinogenicity given that the putative mode of action understanding for chloroform involves cytotoxicity as a key event preceding tumor development” (63 *FR* 15686).

Given the recent evaluations presented by OW that conclude that protecting against chloroform’s noncancer health effects protects against excess cancer risk, EPA now believes that the noncancer health effects resulting from inhalation of chloroform would precede the development of cancer and would occur at lower doses than tumor (cancer) development. Although EPA has not finalized a noncancer health benchmark for inhalation exposure (a reference concentration, RfC), the Agency for Toxic Substances and Disease Registry (ATSDR) has developed a Minimal Risk Level (MRL) for inhalation exposure to chloroform. An MRL is “an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure [acute, intermediate, or chronic]” (http://www.atsdr.cdc.gov/mrls.html). The ATSDR MRLs are intended to serve as screening levels, and are used to identify contaminants and potential health effects that may be of concern at hazardous waste sites. During the development of ATSDR’s toxicological profiles, MRLs are derived when ATSDR determines that reliable and sufficient data exist to identify the target organ(s) of effect, or the most sensitive health effect(s) for a specific duration for a given route of exposure to the substance. MRLs are based on noncancer health effects only. ATSDR uses the no-observed-adverse-effect-level/uncertainty factor approach to derive MRLs for hazardous substances. MRLs are set below levels that, based on current information, might cause adverse health effects in the people most sensitive to such substance-induced effects. MRLs generally are based on the most sensitive substance-induced endpoint considered to be of relevance to humans. Most MRLs contain some degree of uncertainty because of the lack of precise toxicological information on the people who might be the most sensitive (for example, infants, elderly, and the nutritionally or immunologically compromised) to the effects of hazardous substances. ATSDR uses a conservative (that is, protective) approach to address these uncertainties. Proposed MRLs undergo a rigorous review process. They are reviewed by ATSDR’s Health Effects/MRL Workgroup within the Division of Toxicology; an expert panel of external peer reviewers; the ATSDR-wide MRL Workgroup, with participation from other Federal agencies, including EPA; and are submitted for public comment through the toxicological profile public comment period.

To evaluate the noncancer hazard associated with exposure to chloroform in air, we compared the concentration of chloroform that we predicted to occur at a high end receptor's point of exposure to the ATSDR MRLs for inhalation exposure to chloroform. The highest chloroform exposure point concentration in air modeled for chlorinated aliphatics wastewaters was 0.74 ug/m^3 (Table 4-1). This concentration is approximately 0.0001 ppm (assuming 1 atmosphere pressure and 16^o Celsius, the average for the meteorological stations we evaluated), which is more than two orders of magnitude below the chronic inhalation MRL for chloroform, 0.02 ppm (the chronic MRL is more protective than either the acute or intermediate MRLs), indicating that there is no concern for adverse noncancer health effects, or, therefore, significant increased risk of cancer, resulting from inhalation exposure to chloroform emitted from chlorinated aliphatics wastewaters.

Table 4-1. Chloroform Exposure Point Concentrations in Air (Tank, Non-groundwater Deterministic Analysis)

Receptor(s)	Scenario	Air Concentration (ug/m3)
Farmer	Central Tendency	3.34E-02
	High End (Exposure duration, Distance to receptor)	2.13E-01
Adult resident, Gardener, Fisher	Central Tendency	3.34E-02
	High End (Waste concentration, Distance to receptor)	7.42E-01
Child of Resident, Child of Farmer	Central Tendency	3.34E-02
	High End (Waste concentration, Distance to receptor)	7.42E-01

5. RISK CHARACTERIZATION

This section provides technical information that supplements that provided in Section 5 of the 1999 Risk Assessment Technical Background Document. Section 5 of the background document summarized the deterministic and probabilistic risk analyses for our constituents of concern and discussed sources of uncertainty in the risk estimates.

The information we provide in this section includes:

- A discussion of the likelihood of noncancer health effects resulting from exposure to dioxins in chlorinated aliphatics wastes and an evaluation of dioxin exposure for nursing infants; and
- A summary of pathway and exposure route-specific land treatment unit TCDD TEQ risk estimates for the adult farmer; and
- A summary of high end risk estimates for chlorinated aliphatics wastewaters and EDC/VCM sludges.
- A revised population risk estimate for the EDC/VCM land treatment unit.

5.1 Non-Cancer Health Effects Resulting from Exposure to Dioxins and Evaluation of Dioxin Exposure for Nursing Infants

Typically, EPA calculates a hazard quotient (HQ) to assess the noncancer health effects resulting from contaminant exposure. For oral exposures, the HQ is the ratio of an individual's daily contaminant dose to the reference dose (RfD) for the contaminant. EPA has not established RfDs for any of the dioxin or furan congeners (USEPA, 1994). EPA is awaiting the finalization of the Draft Dioxin Reassessment before formalizing an approach to evaluating noncancer risks from dioxin. In recent years EPA's Office of Solid Waste and Emergency Response (OSWER) has calculated a margin of incremental exposure (MOIE) to dioxin on a case-by-case basis as a tool for evaluating the potential for the occurrence of noncancer health effects due to dioxin. The margin of incremental exposure is an expression of the additional (increment of) exposure to dioxin that an individual receives in excess of background exposure to dioxin.

Accordingly, we calculated the incremental dioxin exposure for a breast-feeding infant whose mother's daily intake of dioxin is 12 pg TCDD International (I)-TEQ, the estimated high end dioxin daily intake under the adult farmer scenario evaluated in the proposed wastewater risk analysis (see the revised version of Table 4-4 in the Errata section of this document). The steps for conducting this calculation are described below.

1. Estimate the TCDD TEQ concentration in the mother's body fat. We used the following simple single-compartment, first-order pharmacokinetic model to estimate the TCDD TEQ concentration in body fat as a function of the intake of dioxins:

$$C_{bf} = \frac{d \; h \; CF_1}{0.693 \; V \; CF_2}$$

where

C_{bf} is the TCDD TEQ concentration in body fat (pg/mL fat, or ppt fat)
d is the intake of dioxin (pg/day)
h is the half-life of dioxins in adults (years)
V is the volume of body fat (L)
CF_1 is a conversion factor that converts years to days (days/year)
CF_2 is a conversion factor that converts L to mL (mL/L).

The values for the variables in this equation are as follows:

Variable	Value	Units	Reference
d, intake of dioxin	12	pg/day	Table 4-4, Section 8
h, half-life of dioxins in adults	7.2	years	Flesch-Janys et al. (1996) estimated the half-life of 2,3,7,8-TCDD (and other CDD/CDFs) based on blood levels of a group of occupationally exposed individuals. The median half-life for 2,3,7,8-TCDD (n=48) was estimated to be 7.2 years. The value for 2,3,7,8-TCDD is assumed applicable to the TCDD TEQ.
V, volume of body fat	14	L	Assumption. Assuming 1 L = 1 kg, 14 L is 20% of a 70 kg adult.
CF_1, conversion factor	365	day/yr	
CF_2, conversion factor	1000	mL/L	

Using these values, we estimate the TCDD TEQ concentration in body fat to be 3.25 pg/mL fat. In Step 2, below, we assume that the TCDD TEQ concentration in the mother's body fat is the same as the TCDD TEQ concentration in the mother's breast milk fat. There are a number of other uncertainties in the use of this equation, including our assumptions regarding a 7.2 year half-life for TEQs, the volume of body fat, and the use of a steady state model.

2. Estimate the dioxin dose to the infant. We estimate dose to the infant as:

$$ADD_{infant} = \frac{C_{mf}\, f_3\, f_4\, IR_{milk}\, ED}{BW_{infant}\, AT}$$

where

ADD_{infant} is the dose to the infant (pg/kg-day)

C_{mf} is the contaminant concentration in milk fat (pg/mL), assumed to be the concentration in the mother's body fat

f_3 is the fraction of fat in breast milk (unitless)

f_4 is the fraction of ingested contaminant that is absorbed (unitless)

IR_{milk} is the ingestion rate of milk (mL/day)

ED is the time during with breast feeding occurs, or the exposure duration (yr)

BW_{infant} is the infant body weight (kg)

AT is the averaging time (yr).

The values for the variables in this equation are as follows:

Variable	Value	Units	Reference
$C_{mf,}$ contaminant concentration in milk fat	3.25	pg/g	Assumed to be same as TCDD TEQ concentration in mother's body fat, $C_{bf,}$ calculated in Step 1
$f_{3,}$ fraction of fat in breast milk	0.04	unitless	USEPA, 1997; Table 4-16
$f_{4,}$ fraction of ingested contaminant that is absorbed	0.9	unitless	This assumption is supported by Lakind, et al. (2000), who selected a 95% absorption of 2,3,7,8-TCDD in a nursing model for TCDD and cited four studies showing this much absorption for the lower chlorinated dioxin compounds. Assuming 90% absorption may be conservative for the higher chlorinated dioxins, which are shown to be excreted at a higher rate.
IR_{milk}, ingestion rate of breast milk	688	mL/day	12 month time-weighted average mean breast milk intake; USEPA, 1997; Table 4-16
ED, exposure duration	1	yr	Assumption, period of time infant is breast-fed
$BW_{infant,}$ infant body weight	7.2	kg	Time weighted average of 50th percentile body weights for infants 0-12 months; USEPA, 1997; Table 7-1. Average of values for males and females.
AT, averaging time	1	yr	equivalent to exposure duration

With a TCDD TEQ concentration in body fat of 3.25 pg/g (calculated in Step 1), the dose to the infant is estimated as 11.2 pg/kg-day under the high end wastewater exposure scenario. Compared to the expected infant background dose of 60 pg TCDD I-TEQ/kg-day due to exposure to dioxins and furans[4] (Schaum et al., 1999; USEPA, 1994), an infant's high end indirect exposure to dioxin in chlorinated aliphatics wastewaters represents an incremental exposure of 19 percent of an infant's background exposure. EPA also evaluated adult exposure to dioxin from chlorinated aliphatics wastewaters. Compared to an expected adult daily background dose of 1 pg TCDD World Health Organization (WHO)-TEQ/kg-day[5] due to exposure to dioxins, furans, and coplanar PCBs (Schaum et al., 1999), the incremental dose due to chlorinated aliphatics wastewaters, 0.17 pg TCDD I-TEQ/kg-day, is 17 percent of background. Modifications to EPA's proposed wastewater risk estimates summarized in

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[4] Schaum et al. (1999) report that "[b]ecause PCB TEQ concentrations in breast milk are similar to those of CDD/Fs, adding the PCB dose from breast milk intake approximately doubles the total TEQ dose to infants."

[5] Schaum et al. (1999) report that their results reported as WHO_{98} TEFs are about 10% greater than the same results using the I-TEFs.

Section 5.3 would result in reducing the incremental exposure for both adults and nursing infants to less than 10 percent (about 7 percent) of background exposure.

Under the land treatment scenario, the high end adult farmer has a TCDD TEQ intake of 130 pg/day (Table 4-4, USEPA, 1999; Table 4-4, Section 8, this document), or 1.9 pg/kg-day, which is 186 percent of background exposure. Using the adult's value for intake, and the equations in Steps 1 and 2 above, the estimated dose to the infant is 121.1 pg/kg-day. Compared to the expected infant background dose of 60 pg TCDD I-TEQ/kg-day, an infant's high end indirect exposure to dioxin in EDC/VCM sludges managed in a land treatment unit represents an incremental exposure of 202 percent of an infant's background exposure. Modifications to EPA's proposed EDC/VCM sludge risk analysis summarized in Section 5.3 would result in reducing the incremental exposure for adults to approximately 68 percent of background and, for nursing infants, to about 74 percent of background exposure.

5.2 Pathway and Exposure Route-Specific Land Treatment Unit TCDD TEQ Risk Estimates for the Adult Farmer

In Appendix H.1 of the 1999 Risk Assessment TBD we provided data for chlorinated aliphatic wastewaters managed in aerated biological treatment tanks that allowed the reader to calculate the proportion of the high end deterministic risk for the adult farmer that was due to the various pathways and route of exposure. We provided similar information for the EDC/VCM wastewater treatment sludges managed in a land treatment unit, but not in the same level of detail. Table 5-1 provides a summary of this information for the adult farmer receptor under the land treatment unit scenario. Some of this information is redundant with

Table 5-1. TCDD TEQ Risk Results for the Adult Farmer, Land Treatment Unit

Exposure Route and Pathway		Percentage of High End **Pathway Risk** Attributable to:	Percentage of **Total Risk** Attributable to:	
			Central Tendency	**High End**
Vegetables (Root and Aboveground)				
	Erosion and Runoff	63.6	0.3	0.5
	Air (Total)	36.4	0.2	0.1
	Vapor	33.3		
	Particulate	3.1		
	Sum		0.5	**0.6**
Fruit				
	Erosion and Runoff	65.3	0.72	0.72
	Air (Total)	34.7	0.39	0.39
	Vapor	33.5		
	Particulate	1.2		
	Sum		1.1	**1.1**
Soil				
	Erosion and Runoff	98.0	2.2	2.4
	Air (Total)	2.0	0.05	0.05
	Vapor	1.2		
	Particulate	0.8		
	Sum		2.2	**2.4**
Dairy				
	Erosion and Runoff	29.3	13.0	13.1
	Air (Total)	70.7	31.8	32.0
	Vapor	69.4		
	Particulate	1.3		
	Sum		44.8	**45.1**
	Forage	72.6		
	Grain	0.5		
	Silage	0.6		
	Soil	26.3		
Beef				
	Erosion and Runoff	42.0	21.6	21.4
	Air (Total)	58.0	29.9	29.5
	Vapor	56.9		
	Particulate	1.1		
	Sum		51.5	**50.9**
	Forage	59.3		
	Grain	0.1		
	Silage	0.5		
	Soil	40.2		

the data provided in Table 5-8 of the 1999 Risk Assessment TBD, but is presented again for the convenience of the reader.

5.3 Summary of High End Risk Estimates – Chlorinated Aliphatics Wastewaters and EDC/VCM Sludges

This section summarizes the how the evaluations and conclusions presented in the previous sections would influence the risk estimates that we presented in the 1999 Risk Assessment TBD. Because our listing determinations for chlorinated aliphatic wastewaters and EDC/VCM sludges managed in a land treatment unit were based primarily on the high end deterministic dioxin risks for the adult farmer (expressed as TCDD TEQ), and because these are the highest risks that we identified, the sections below focus on how the factors that reduce our risk estimates, as highlighted in the previous sections, would influence the deterministic high end adult farmer dioxin risks.

5.3.1 Chlorinated Aliphatic Wastewaters Managed in Aerated Biological Treatment Tanks

As a result of our analysis of comments provided on the Agency's Proposed Rule for chlorinated aliphatics wastes (64 FR 46476), we determined that it was appropriate to adjust our chlorinated aliphatic wastewater risk assessment results to account for three factors that were not addressed in the July 1999 risk assessment, and that would reduce our high end deterministic risk estimate for the adult farmer. These factors are discussed in previous sections of this document, and are as follows:

- Modifying the assumptions regarding the size of the pasture on which the cattle graze.
- Accounting for cooking and post-cooking loss of beef.
- Assuming that solids are removed from the wastewater prior to aerated biological treatment

These modifications, and their impact on the proposed risk estimate, are summarized on the following table (Table 5-2).

Table 5-2. Modifications to the High End Deterministic Risk Estimate (TCDD TEQ) for the Adult Farmer – Chlorinated Aliphatics Wastewaters

Risk Estimate presented in 1999 Risk Assessment TBD:	2E-05	
Modification:	New Risk Estimate*:	
Assuming farmer only raises beef and dairy cattle	2E-05	
Assuming farmer does not provide cattle home-produced grain and silage	2E-05	
Accounting for cooking and post-cooking loss of beef	1E-05	
Modifying assumptions regarding the pasture size	7E-06	
Assuming that solids are removed from the wastewater prior to aerated biological treatment	70% Removal	50% Removal
	4E-06	6E-06

* Each new risk estimate includes the reduction in risk presented in the previous row of the table.

As discussed in Section 4.3, EPA no longer believes there are risks due to exposure to chloroform derived from chlorinated aliphatics wastewaters. For the adult farmer receptor, the sum of the inhalation and ingestion deterministic high end cancer risk estimates for the chlorinated aliphatic wastewater COPCs ranges from 5E-06 to 7E-06, depending on the amount of solids removal that occurs prior to aerated biological treatment[6] (5E-06 corresponds to 70% solids removal and 7E-06 corresponds to 50% solids removal). The sum of the inhalation and ingestion deterministic high end noncancer hazard estimates for the chlorinated aliphatic wastewater COPCs is equal to 0.04. These summed estimates are likely overestimates because: 1) they are sums of separate high end risk and hazard estimates that may be based on different high end parameters, thus a number of high end parameters may be combined into the summed estimate; 2) the cooking and post-cooking loss correction for beef intakes has only been made for dioxins and furans; 3) the solids removal correction has only been made for dioxins and furans for ingestion exposures (this correction would be most important for

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[6] We only calculated the effect of solids removal for ingestion exposures to the dioxins and furans. The sum of the high end deterministic inhalation risk estimates for the COPCs is 9E-07 (including an unadjusted dioxin inhalation risk estimate), and the sum of the high end deterministic ingestion risk estimates for the COPCs other than dioxins/furans is 2E-09.

hydrophobic contaminants); and 4) the pasture/field size correction has only been made for dioxins and furans.

5.3.2 EDC/VCM Sludges

As a result of our analysis of comments provided on the Agency's Proposed Rule for chlorinated aliphatics wastes (64 FR 46476), we determined that it was appropriate to adjust our land treatment unit risk assessment results to account for three factors that were not addressed in the July 1999 risk assessment, and that would reduce our high end deterministic risk estimate for the adult farmer. These factors are discussed in previous sections of this document, and are as follows:

- Accounting for cooking and post-cooking loss of beef.
- Applying a mass balance correction for the erosion pathway.
- Modifying the assumptions regarding the size of the pasture on which the cattle graze.

These modifications, and their impact on the proposed risk estimate, are summarized on the following table (Table 5-3).

Table 5-3. Modifications to the High End Deterministic Risk Estimate (TCDD TEQ) for the Adult Farmer – EDC/VCM Sludges Managed in a Land Treatment Unit

Risk Estimate presented in 1999 Risk Assessment TBD:	2E-04
Modification:	New Risk Estimate*:
Assuming farmer only raises beef and dairy cattle	2E-04
Assuming farmer does not provide cattle home-produced grain and silage	2E-04
Accounting for cooking and post-cooking loss of beef	1E-04
Applying a mass balance correction for the erosion pathway	1E-04
Modifying assumptions regarding the pasture size	7E-05

* Each new risk estimate includes the reduction in risk presented in the previous row of the table.

For the EDC/VCM landfill nongroundwater pathway analysis, the highest risk estimates and hazard quotients are associated with the adult farmer receptor. As presented in the 1999 Risk Assessment TBD, the sum of the high end ingestion risk estimates for the adult farmer is 7E-10 and the sum of the high end inhalation risk estimates for the adult farmer is 1E-10, for a total nongroundwater pathway risk estimate of 8E-10. Nongroundwater pathway hazard quotients are less than 0.0001 for all contaminants. Our characterization of the high end groundwater pathway arsenic risk estimate for the farmer, 3E-05 is discussed in the preamble to the final rule. The sum of all other groundwater pathway risk estimates, regardless of the timeframe in which the they occur, is only 1E-06. Similarly, the sum of all groundwater pathway hazard quotients, regardless of the timeframe in which they occur, is only 0.3. Moreover, the predicted risks for the nongroundwater pathway tend to be overestimates because of the pasture size and beef ingestion rate assumed in the analysis (described in the preamble to the final rule), as well as for the reasons described on pages 5-5 and 5-7, and in Appendix H.3, of the 1999 Risk Assessment TBD. An evaluation of public and peer review comment on the proposed rule provided no basis for increasing these estimates.

5.3.3 Probabilistic Risk Results

Applying the modifications discussed in the previous sections to our probabilistic risk estimates is somewhat problematic since the results reported for any one percentile of the probabilistic distribution are not based on one set of data that can be adjusted in the same way as we adjusted the deterministic results. However, we assume that there will be the same general proportion of risk reduction for the probabilistic results as we calculated above for the deterministic results. In addition, as previously mentioned in Section 2.3, the probabilistic risk results are probably slightly higher than they would have been if we had evaluated receptor distances from the source that were more equally distributed.

5.4 Population Risk Estimates

The population risk estimate for the EDC/VCM land treatment unit that EPA presented in the 1999 Risk Assessment TBD (Section 5.2) was based on an adult beef intake rate that did not include cooking and post-cooking losses of beef. Because the population risk estimate was based solely on beef intake, factoring cooking and post-cooking losses of beef into the population risk estimate would modify the population risk estimate by a factor of 0.55, reducing the estimate to 9E-05.

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7. APPENDICES

The following sections present revised versions of four of the Appendices that were included in the 1999 Risk Assessment TBD. These appendices were revised in response to comments from peer reviewers who believed that the appendices required additional clarity. The appendices were not revised to reflect any new analyses or changes in the Agency's methodologies that occurred since the 1999 Risk Assessment TBD was completed, only to provide the clarity and additional detail requested by the peer reviewers. The revised appendices presented in the following sections are as follows:

- Appendix D.1, Source Term Partitioning Model;
- Appendix D.2, Overland Transport Model;
- Appendix D.4, Groundwater Fate and Transport Modeling Using EPACMTP;
- Appendix D.5, Surface Water Model; and
- Appendix F.1, Monte Carlo Fate and Transport Modeling.

Appendix D.1

Source Term Partitioning Model

Appendix D.1

Source Term Partitioning Model

This appendix describes the source term partitioning models that were used to estimate contaminant losses for the following waste management scenarios:

- EDC/VCM sludge managed in onsite land treatment unit (LTU);
- EDC/VCM sludge managed in offsite municipal landfill; and
- Methyl chloride sludge managed in onsite, nonhazardous landfill.

The following discussion describes the general model construct developed to partition chemical constituents in the waste management unit, and then provides a description of how the general model was applied to the LTU and landfills, respectively.

Partitioning Model

The source term partitioning model is a time-dependent model that estimates the contaminant losses from a contaminant source zone (e.g., LTU or landfill) due to volatilization, leaching, rainwater runoff, and chemical transformation (e.g., biodegradation and hydrolysis). The model assumes linear partitioning and first order rate losses and uses a finite difference (numerical) integration approach to solve the mass balance equations. This spreadsheet model uses the Jury solution (Jury et al., 1983, 1984, and 1990) as the basis for the volatilization flux calculation. As implemented, the model calculates the average annual contaminant source zone concentration, the annual mass of contaminant volatilized, and the annual mass of contaminant removed in the leachate.

Model Theory

Equilibrium Partitioning

The total concentration of contaminant in the source (i.e., waste unit) can be expressed as the sum of the concentrations of contaminant adsorbed on the soil or waste particles, dissolved in the liquid, and in the air spaces as follows:

$$C_T = C_s + (\theta_w C_w/\rho_b) + (\theta_a C_a/\rho_b) \tag{1}$$

where

C_T = total contaminant concentration (mg/kg = g/Mg)
C_s = concentration of contaminant adsorbed on soil (mg/kg = g/Mg)
θ_w = water-filled soil porosity (m^3_{water}/m^3_{soil})
C_w = concentration of contaminant in liquid ($\mu g/cm^3 = g/m^3$)
ρ_b = soil dry bulk density ($g/cm^3 = Mg/m^3$)
θ_a = air-filled soil porosity (m^3_{air}/m^3_{soil})
C_a = concentration of contaminant in air ($\mu g/cm^3 = g/m^3$).

The adsorbed contaminant concentration is assumed to be linearly related to the liquid phase concentration as follows:

$$C_s = K_d C_w \tag{2}$$

where

C_s = concentration of contaminant adsorbed on soil (mg/kg = g/Mg)
K_d =soil-water partition coefficient ($cm^3/g = m^3/Mg$) = $K_{oc} f_{oc}$ for organic compounds
K_{oc} = soil organic carbon partition coefficient (cm^3/g)
f_{oc} = organic carbon content of soil (g/g)

C_w = concentration of contaminant in liquid ($\mu g/cm^3 = g/m^3$).

The contaminant concentration in the vapor phase is assumed to be linearly related to the liquid phase concentration as follows:

$$C_a = H' C_w \tag{3}$$

where

C_a = concentration of contaminant in air ($\mu g/cm^3 = g/m^3$)
H' = dimensionless Henry's law constant = 41 × H
H = Henry's law constant at 25°C ($atm\text{-}m^3/mol$)
C_w = concentration of contaminant in liquid ($\mu g/cm^3 = g/m^3$).

Equations 2 and 3 assume linear equilibrium partitioning between the adsorbed contaminant, the dissolved contaminant, and the volatilized contaminant. Combining Equations 1, 2, and 3 yields:

$$C_T = C_s\,[1 + \theta_w/(K_d\rho_b) + \theta_a\,H'/(K_d\rho_b)]. \tag{4}$$

The total contaminant concentration, C_T, represents the measured soil concentration. However, it is the adsorbed soil concentration that is needed to calculate the equilibrium liquid and air contaminant concentrations (Equations 2 and 3). Equation 4 can be rearranged to calculate the adsorbed soil contaminant concentration given the total contaminant concentration as follows:

$$C_s = \frac{C_T \ K_d \ \rho_b}{(K_d \ \rho_b \ - \ \theta_w \ - \ \theta_a \ H')} \ . \qquad (5)$$

Overall Mass Balance

For a constant volume system assuming first-order rate loss mechanisms, the mass balance can be expressed as:

$$(\rho C_T \ / \ \rho t) = - (k_{app,air} + k_{app,leach} + k_{app,runoff} + k_{app,chemt}) \ C_T \qquad (6)$$

where

$k_{app,air}$ = the apparent first order rate constant for volatilization, 1/s

$k_{app,leach}$ = the apparent first order rate constant for leaching, 1/s

$k_{app,runoff}$ = the apparent first order rate constant for rain runoff, 1/s

$k_{app,chemt}$ = the apparent first order rate constant for chemical transformation, 1/s

C_T = total contaminant concentration (mg/kg = g/Mg).

[Note that soil erosion may also lead to contaminant mass losses; however, soil erosion was not considered as a loss mechanism in the source partitioning model.]

For small enough time steps (time steps in which C_T changes only a few percent), Equation 6 can be approximated as follows:

$$(M_{s,t+{}^\circ t} - M_{s,t})/({}^\circ t) = - (k_{app,air} + k_{app,leach} + k_{app,runoff} + k_{app,chemt}) \ M_{s,t} \qquad (7)$$

or

$${}^\circ M_{tot} = {}^\circ M_{air} + {}^\circ M_{leach} + {}^\circ M_{runoff} + {}^\circ M_{chemt} \qquad (8)$$

where

$M_{s,t+{^\circ}t}$ = mass of contaminant in soil at time t+°t (g)

$M_{s,t}$ = mass of contaminant in soil at time t (g)

${^\circ}t$ = time step of calculation (s)

$k_{app,air}$ = the apparent first order rate constant for volatilization (1/s)

$k_{app,leach}$ = the apparent first order rate constant for leaching (1/s)

$k_{app,runoff}$ = the apparent first order rate constant for rain runoff (1/s)

${^\circ}M_{tot}$ = total mass of contaminant removed from soil over time step (g) = $M_{s,t} - M_{s,t+{^\circ}t}$

${^\circ}M_{air}$ = mass of contaminant lost over time step due to volatilization (g)

${^\circ}M_{leach}$ = mass of contaminant lost over time step due to leaching (g)

${^\circ}M_{runoff}$ = mass of contaminant lost over time step due to runoff (g)

${^\circ}M_{chemt}$ = mass of contaminant lost over time step due to chemical transformation (g).

Due to the simplified nature of the numerical integration used, any number of competing loss mechanisms can be included in the model because each of the loss mechanisms can be evaluated separately and summed together.

Loss to Atmosphere

The primary mechanism of contaminant loss to the atmosphere is the diffusion of volatilized contaminant to the soil surface. During periods of evaporation, the flux of water vapor enhances contaminant transport to the soil surface. Consequently, the total contaminant flux to the atmosphere is:

$$J_{air,t} = J_{vol,t} + J_{evaptr,t} \tag{9}$$

where

$J_{vol,t}$ = contaminant flux to the atmosphere due to diffusion (g/m^2-s)
$J_{evaptr,t}$ = contaminant flux to the atmosphere due to evaporative transport (g/m^2-s).

The total mass loss to the air can be calculated as follows:

$$^{\circ}M_{air} = (J_{evaptr,t} + J_{vol,t})(A\,^{\circ}t). \tag{10}$$

where

$J_{evaptr,t}$ = contaminant flux to the atmosphere due to evaporative transport (g/m^2-s)
$J_{vol,t}$ = contaminant flux to the atmosphere due to diffusion (g/m^2-s)
A = area of contaminant source (m^2)
$^{\circ}t$ = time step of calculation (s).

The contaminant flux to the atmosphere terms are estimated using the following equations based on whether the contaminated soil layer is in direct contact with the atmosphere ("no soil cover") or buried beneath a layer of uncontaminated soil ("soil cover").

<u>Emissions With No Soil Cover</u>

Assuming that there is no soil cover and no stagnant boundary air layer at the ground surface, the Jury et al. (1990) simplified finite source model for diffusional volatilization can be written as:

$$J_{vol,t} = C_T \left(\frac{0.01\ D_A}{\rho t} \right)^{½} \left[1 - \exp\left(\frac{-d_s^2}{0.04\ D_A\ t} \right) \right] \tag{11}$$

where

C_T = total contaminant concentration (mg/kg = g/Mg)

D_A = apparent diffusivity (cm^2/s)

ρ = 3.14

t = time (s)

d_s = depth of uniform soil contamination at t=0, i.e., depth of daily addition (m).

and

$$D_A = \left[\frac{\left(\theta_a^{\frac{10}{3}} \; D_i \; H' \; - \; \theta_w^{\frac{10}{3}} D_w \right)}{n^2 \; (\rho_b \; K_d \; - \; \theta_w \; - \; \theta_a \; H')} \right] \qquad (12)$$

where

θ_a = air-filled soil porosity (m$^3_{\text{air}}$/m$^3_{\text{soil}}$)

D_i = diffusivity in air (cm^2/s)

H' = dimensionless Henry's law constant = 41 × H

H = Henry's law constant at 25°C (atm-m^3/mol)

θ_w = water-filled soil porosity (m$^3_{\text{water}}$/m$^3_{\text{soil}}$)

D_w = diffusivity in water (cm^2/s)

n = total soil porosity ($L_{\text{pore}}/L_{\text{soil}}$) = 1 - ($\rho_b/\rho_s$)

ρ_b = soil dry bulk density (g/cm^3 = Mg/m^3)

ρ_s = soil particle density (g/cm^3)

K_d = soil-water partition coefficient (cm^3/g = m^3/Mg) = K_{oc} f_{oc} for organic compounds

K_{oc} = soil organic carbon partition coefficient (cm^3/g)

f_{oc} = organic carbon content of soil (g/g).

As discussed in Jury et al. (1984), volatilization with evaporation is a complex problem, but evaporation always increases the overall volatilization rate. Jury et al. (1984) presents an equation for the convection of contaminants caused by the flux of water in the soil. The convective volatilization flux caused by evaporation is calculated by isolating the first half of the overall volatilization flux equation (Jury et al., 1983), which can be written as follows:

$$J_{evaptr,t} = ½\ C_T\ \rho_b\ (0.01\ V_E) \left[erfc\left(\frac{V_E\ t}{(4\ D_A\ t)^{½}} \right) - erfc\left(\frac{(100\ d_s\ -\ V_E\ t)}{(4\ D_A\ t)^{½}} \right) \right] \tag{13}$$

where

C_T = total contaminant concentration (mg/kg = g/Mg)

ρ_b = soil dry bulk density (g/cm^3 = Mg/m^3)

V_E = evaporative convective velocity (cm/s)

erfc(x) = complementary error function

t = time (s)

D_A = apparent diffusivity (cm^2/s)

d_s = depth of uniform soil contamination at t=0 (m);

and

$$V_E = \left(\frac{-E}{(365\times24\times3600)\ \times\ (\rho_b\ K_d\ -\ \theta_w\ -\ \theta_a\ H')} \right) \tag{14}$$

where

E = average annual evaporation rate (cm/yr)

ρ_b = soil dry bulk density (g/cm^3 = Mg/m^3)

K_d =soil-water partition coefficient (cm^3/g = m^3/Mg) = K_{oc} f_{oc} for organic compounds

K_{oc} = soil organic carbon partition coefficient (cm^3/g)

f_{oc} = organic carbon content of soil (g/g)

θ_w = water-filled soil porosity (m^3_{water}/m^3_{soil})

θ_a = air-filled soil porosity (m^3_{air}/m^3_{soil})

H' = dimensionless Henry's law constant = 41 × H

H = Henry's law constant at 25°C (atm-m^3/mol)

[Note: The minus sign is introduced because upward movement is in the negative direction.]

Mass Lost Via Leaching

Leachate concentrations are calculated by dividing the mass of contaminant lost via leaching by the volume of leachate generated per year (which is the annual infiltration rate times the area of the waste management unit).

The mass flux loss of a contaminant due to leaching is estimated by assuming the leachate is in equilibrium with the soil (i.e., Equation 2 applies).

$$J_{leach,t} = \frac{C_T \ \rho_b \ (0.01 \ V_L)}{(\rho_b \ K_d - \theta_w - \theta_a \ H')} \tag{15}$$

where

$J_{leach,t}$ = contaminant flux in leachate at time t, g/m^2-s

C_T = total contaminant concentration (mg/kg = g/Mg)

ρ_b = soil dry bulk density (g/cm^3 = Mg/m^3)

V_L = (P + I - R - E)/(365 × 24 × 3600] = leachate rate (cm/s)

P = annual average precipitation rate (cm/yr)

I = annual average irrigation rate (cm/yr)

R = annual average runoff rate (cm/yr)

E = average annual evaporation rate (cm/yr)

K_d = soil-water partition coefficient ($cm^3/g = m^3/Mg$) = $K_{oc}\ f_{oc}$ for organic compounds

K_{oc} = soil organic carbon partition coefficient (cm^3/g)

f_{oc} = organic carbon content of soil (g/g)

θ_w = water-filled soil porosity (m^3_{water}/m^3_{soil})

θ_a = air-filled soil porosity (m^3_{air}/m^3_{soil})

H' = dimensionless Henry's law constant = 41 × H

H = Henry's law constant at 25°C (atm-m^3/mol)

In the same fashion that the air fluxes were converted a mass loss, the leaching flux rate can be converted to a mass loss as follows:

$$^{\circ}M_{leach} = (J_{leach,t})(A^{\circ}t). \tag{16}$$

Loss Due to Runoff

The equation describing the mass flux loss of a contaminant due to runoff is nearly identical to Equation 15, because the runoff is also assumed to be in equilibrium with the contaminated soil.[7] Consequently, the total mass rate of contaminant loss due to runoff is

$$J_{runoff,t} = \frac{C_T\ \rho_b\ (0.01\ V_R)}{(\rho_b\ K_d - \theta_w - \theta_a\ H')} \tag{17}$$

where

$J_{runoff,t}$ = contaminant runoff rate at time t (g/m^2-s)

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[7] During the development of the model, we reviewed the assumption that runoff is in equilibrium with the soil. Because data on storm events and rain intensity would be required to evaluate nonequilibrium runoff, we did not attempt to define the proportion of runoff that enters the soil (versus that which does not). The runoff that causes soil erosion would mix well with the eroding soil particles, some of which may be redeposited on the LTU. Consequently, even the runoff that does not enter the soil will contain some contaminant. Given the mathematical construct used to estimate contaminant partitioning in soil, and data availability on precipitation, evaporation, and runoff, assuming equilibrium conditions was considered the most technically defensible option.

C_T = total contaminant concentration (mg/kg = g/Mg)

ρ_b = soil dry bulk density (g/cm^3 = Mg/m^3)

V_R = R/(365 × 24 × 3600] = runoff rate (cm/s)

K_d =soil-water partition coefficient (cm^3/g = m^3/Mg) = K_{oc} f_{oc} for organic compounds

K_{oc} = soil organic carbon partition coefficient (cm^3/g)

f_{oc} = organic carbon content of soil (g/g)

θ_w = water-filled soil porosity (m^3_{water}/m^3_{soil})

θ_a = air-filled soil porosity (m^3_{air}/m^3_{soil})

H' = dimensionless Henry's law constant = 41 × H

H = Henry's law constant at 25°C (atm-m^3/mol)

Then,

$$^{\circ}M_{runoff} = (J_{runoff,t})(A\,^{\circ}t). \qquad (18)$$

Loss Due to Chemical Transformation

Chemical transformation (e.g., biodegradation and hydrolysis) rates were imputed from reported contaminant half-lives in soil. The contaminant half-life was used to calculate an overall first order rate constant that included all loss mechanisms. Then the total mass loss predicted from this overall first order rate constant was calculated as follows:

$$^{\circ}M_{tot} = M_{s,t}\,[1 - \exp(-\,k_{app,overall}\ ^{\circ}t)] \qquad (19)$$

where

$^{\circ}M_{tot}$ = total mass of contaminant loss from the system (g)

$M_{s,t}$ = mass of contaminant in soil at time t (g)

$k_{app,overall}$ = overall apparent first order rate constant for contaminant loss by all mechanisms (1/s)

$^{\circ}t$ = time step of calculation (s).

If ρM_{tot} exceeds the mass losses from volatilization, leaching and runoff, then the additional mass loss was attributed to (biodegradation+hydrolysis). If the volatilization, leaching and runoff mass losses exceeded ρM_{tot}, then the mass chemically transformed was set to zero.

Application of Model to Land Treatment Unit

In applying the general model to the LTU, we made certain simplifying assumptions. First, we assumed the contaminant source zone is a thoroughly mixed system and, therefore, a concentration profile is not developed over the depth of the zone. Second, we assumed that the contaminant losses due to erosion of soil particles could be neglected without serious implications with regard to mass balance of contaminant. Although it has been shown that organic contaminants tend to sorb to soil fines, and that fine particles of soil tend to erode readily, we accept this limitation and acknowledge that there may be a slight overestimation of the leachate and atmospheric flux because contaminant losses from eroded soil particles are not explicitly accounted for.

Under this waste management scenario, we assumed that the LTU operates for 40 years with annual waste applications (i.e., one application per year). At the end of 40 years, the LTU ceases to receive EDC/VCM sludge, but it may potentially continue to release contaminants into the environment. Therefore, the model tracks the average annual soil concentration, leachate concentration, and the annual mass of contaminant volatilized for the 40 years of active use as well as the period of 40 years of inactive use after the LTU ceases operations (i.e., no additional waste applications). The maximum nine-year leachate concentration predicted by the model is delivered to the groundwater model and used as the starting leachate concentration for the groundwater simulation. The post-closure period of 40 years was used to ensure that the maximum nine-year average leachate concentration is captured over the simulation.

Land treating often involves tilling the waste into the soil. We assumed that the EDC/VCM sludge is tilled to a depth of 0.2 m (U.S. EPA, 1990). We also assumed that there were no runoff/runon controls in place at the LTU to mitigate the release of contaminants via runoff.

The total mass of contaminant applied to the soil during the first annual application can be calculated as follows:

$$M_{s,app} = (C_T\ Q_{app}) \times 1\text{-yr} \tag{20}$$

where

$M_{s,app}$ = mass of contaminant in soil from waste application (g)
C_T = total contaminant concentration (mg/kg = g/Mg)
Q_{app} = annual waste application rate (Mg/yr).

Modeling time-steps for the LTU were set equal to 1 day. After each time step, the mass of constituent remaining in the soil was calculated. We assumed that the contaminant concentrations were uniform over the tilling depth at the beginning of each time step. The model does not generate source zone concentration profiles over time (as a function of depth) given our assumption that active land treatment units are tilled regularly.

Mass additions to the system occur during waste application. The depth of material added during an application is generally negligible; however, some model scenarios could have significant waste material accumulation over 40 years depending on the tilling depth, application rate, and other factors. As waste is sequentially added, and given a fixed tilling depth, a small layer of contaminated soil at the bottom of the land treatment unit will not be tilled in with the newly added waste. The thickness of this layer is equal to the effective thickness of the applied waste (i.e., the volume of the applied waste divided by the surface area of the unit). This layer at the bottom of the unit basically becomes "buried" with successive waste applications and removed from the modeled system. The contaminant remaining in this buried soil is assumed to have negligible effect on the emissions and leachate losses and is effectively removed from the active land treatment unit during waste application.[8] Consequently, the net mass of contaminant added to the land treatment unit, accounting for this "burial loss," at the start of Year 2 through Year 40 is

$$M_{s,app} = C_T\, Q_{app}\, [1 - \{(Q_{app} \times 1\text{-yr})/(A\, \rho_b\,)\}/d_{till}] \times 1\text{-yr} \qquad (21)$$

where

$M_{s,app}$ = mass of contaminant in soil from waste application (g)
C_T = total contaminant concentration (mg/kg = g/Mg)
Q_{app} = annual waste application rate (Mg/yr)
A = area of contaminant source (m^2)

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[8] At the time of burial, the "buried" waste contaminant concentration is less than (or at most equal to) the contaminant concentration in the active 0.2m of the LTU due to contaminant depletion. As such, leachate from the soil in the active 0.2-m of the LTU will be at a higher equilibrium concentration than leachate from the soil in equilibrium with the "buried" contaminated soil layer. Therefore, as described previously, this "buried" or "lost" contaminant will not significantly impact the maximum 9-yr average groundwater concentration at the receptor well. We also note that contaminant burial tends to be a significant contaminant removal mechanism only when other contaminant removal mechanisms are essentially zero (i.e., chemicals that do not degrade, volatilize, or leach).

ρ_b = soil dry bulk density ($g/cm^3 = Mg/m^3$)

d_{till} = tilling depth = 0.2 m.

Application of Model to Landfills

In applying the general partitioning model to the landfill, we make certain simplifying assumptions. First, although the closed landfill cells consist of two layers (i.e., a waste zone and cap zone), application of the Jury solution to predict volatilization from the landfill requires that the soil properties are homogenous in both zones.[9] Consequently, the model does not recognize differences in permeability between the two layers (as a function of density, void fraction, organic content). The contaminant source model does, however, account for the difference in concentrations between the waste zone and cap zone (e.g., the initial contaminant concentration in the cap zone is set to zero). In addition, the source partitioning model *does* produce a concentration profile over depth (one of the benefits of applying the Jury solution). However, the contaminant fluxes are based on total mass in the contaminant source zone and, therefore, the concentration profile is not required by the air dispersion model. Second, the mass of constituent predicted to volatilize by the source partition model was assumed to reduce the total mass of constituent in the landfill available for leaching. Therefore, the total mass volatilized is subtracted from the total mass of constituent placed in the landfill, and this reduced mass is assumed to be the mass of constituent that may be depleted during the life of the simulation (due to leaching). Lastly, no hydrolysis or biodegradation is assumed to occur in the landfill.

Under this waste management scenario, the model was used to evaluate volatilization from the landfill for three conditions. The first condition is the daily waste addition in which the waste is in direct contact with the atmosphere. The second condition is the active landfill cell in

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[9] As indicated in the discussion of the general model, the calculations for volatilization fluxes are based on the Jury solution.

which the waste is covered by a thin "daily" cover. The third condition is the closed landfill cell in which the waste is covered by a thick landfill cap. The source partitioning model was used to estimate the mass of constituent volatilized from the waste, as well as the mass of constituent that leaches from the contaminant source zone. However, the groundwater modeling simulation of the landfill did not use the leachate flux predicted by the source partitioning model. Instead, the landfill leachate concentration from the Toxicity Characteristic Leaching Procedure (TCLP) data was used as the starting leachate concentration.[10] This approach is consistent with the source partitioning model except that TCLP data were to determine the initial leachate concentration rather than deriving an initial leachate concentration using a partition coefficient that describes the relationship between total waste and liquid-phase concentration (as described by Equation 2). In short, the leachate flux was included in the source partition model to account for the fact that leaching may be a competitive removal mechanism to volatilization for certain chemicals.

For the chlorinated aliphatics listing determination, two landfill scenarios were modeled.

- EDC/VCM Sludge Offsite Municipal Landfill - We assumed that EDC/VCM sludges are disposed in offsite in unlined municipal landfills. Based on municipal solid waste management requirements (40 CFR Part 258), we assumed that the landfills are covered daily (every 12 hours) with soil and are capped at the end of their active life, which is 30 years (30 years is the average active lifetime of municipal Subtitle D landfills based on a survey conducted by EPA [U.S.EPA 1988]). We assumed that the landfill continues to release contaminants into the environment for 40 years after it is closed.

- Methyl Chloride Sludge Onsite Nonhazardous Landfill - Review of the §3007 survey responses found that methyl chloride sludge is managed at a single facility in

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[10] For municipal landfill analyses, TCLP data are used in preference to modeled leachate data as a matter of U.S. EPA policy.

> a landfill that is lined with a 24-inch clay liner and has a leachate collection system. We estimated that the landfill would have an active life of 90 years. We assumed that the landfill continues to release contaminants into the environment for 40 years after it is closed.

The waste added to the landfill was assumed to be homogeneous with respect to the concentration throughout the waste as well as the concentration in each waste addition over time. One landfill cell was assumed to be filled per year. The user enters the annual waste quantity and the contaminant concentration of the waste of interest, the waste density, the dimensions of the entire landfill, and the life expectancy of the landfill. From this information, the dilution effect of the target waste being added to other waste in the landfill was calculated. The partitioning equations were applied in three distinct compartments to the landfill model. These included the daily waste addition in which the waste is in direct contact with the atmosphere, the active landfill cell in which the waste is covered by a thin "daily" cover; and the closed landfill cell in which the waste is covered by a thick landfill cap.

Losses from the active landfill cell were estimated by simulating the addition of waste to the landfill on a daily basis. It was assumed that a daily waste addition was 2.5 feet deep and its volume was determined by the annual waste disposal rate, assuming 350 operating days per year. The waste was assumed to be uncovered in the landfill for a user-specified time (model runs were made using an uncovered duration of 12 hours). After the initial uncovered 12 hours, a 6-inch thick cover was added to the waste daily. It was assumed that the daily waste additions were disposed of so that the landfill was filled in layers, with the entire first layer of the cell being filled first, then the second layer was filled in the same order as the first layer, and so on. The total amount of contaminant emitted and leached from the landfill cell was calculated by modeling these individual daily waste additions and summing the results for each waste addition.

After an active landfill cell was completely filled (it was assumed that it took 1 year to fill each cell), the landfill cell was capped. Losses from the entire landfill were estimated by simulating the sequential filling of landfill cells. For example, in Year 1, there were losses from an active cell. In Year 2, there were losses from an active cell plus losses from the first year of a capped cell. In Year 3, there were losses from an active cell plus losses from two capped cells, and so on. Once the entire landfill was filled, the model simulated 30 closed cells from Year 31 to the end of the simulation period.

As indicated above, the landfill leaching losses were estimated using a source partitioning module inside the EPACMTP subsurface fate and transport model (see Appendix D.4 for additional details on the groundwater modeling). The TCLP test is designed to simulate landfill leachate and use of these measured values was deemed appropriate to model the landfill leaching process. Using both the measured waste concentration (C_T) and TCLP leachate concentration (C_L^0), the time-dependent leachate concentration, $C_L(t)$, is given by:

$$C_L(t) = C_L^0 \, e^{-\frac{V_L}{d \, F_h \, \rho_b \, (C_T^0/C_L^0)} t} \tag{22}$$

where

d = Landfill depth (m)
F_h = Volume fraction of the landfill occupied by the waste
C_L^0 = Initial leachate concentration (equal to measured TCLP concentration) (mg/L)
C_T^0 = Initial total waste concentration (mg/kg)
t = time (yr)

This equation does not set a leaching period *a priori*; rather, the leaching process continues until all constituent mass in the landfill has been depleted. During the leaching process, the constituent concentration remaining in the landfill is gradually depleted, and the resulting leachate concentration also diminishes with time following the exponential relationship expressed by Equation 22 (USEPA, 1996). The superscript ^{0} in this equation is used to denote the initial concentration in the waste at the beginning of the leaching process (C_T^0), and the initial leachate concentration, C_L^0, where the latter is given by the measured TCLP concentration. In order to ensure a correct accounting for the total amount of constituent disposed in the landfill, as well as the mass lost via volatilization, the assumption was made that leaching does not begin until the landfill is closed, i.e, F_h in Equation 22 represents the amount of waste when the landfill is “full.” The initial waste concentration, C_T^0 is determined from the waste concentration entering the landfill after adjusting for the mass lost due to volatilization during the landfill active life. As noted above, the source partitioning model was used to calculate the mass lost to volatilization. In modeling the landfill leaching process, additional volatilization losses that may occur after landfill closure are ignored based on the assumption that volatilization losses will be minor following placement of the final landfill cap. This assumption may overestimate the amount of constituent which is available for leaching as it ignores volatilization losses that occur after landfill closure. Note that for the purposes of determining exposures and risk via the atmospheric pathway described previously, volatilization losses for 40 years following closure were modeled.

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Appendix D.2

Overland Transport Model

Appendix D.2

Overland Transport Model

Conceptual Approach to Modeling Overland Transport

The overland transport of contaminants was considered a potentially important mechanism by which constituents in chlorinated aliphatics wastes may be released from a land treatment unit (LTU), and contaminate environmental media and food to which humans may be exposed. Consequently, an overland transport model was developed to predict the erosion losses from the LTU and the resulting contaminant load to a receiving waterbody and the intervening area between the LTU and the waterbody. The Universal Soil Loss Equation (USLE) was chosen as the construct to predict erosion losses from the LTU. The USLE is an empirical erosion model and constitutes a standard tool for exposure assessment involving overland pathways. This empirical model was developed to estimate long-term average soil erosion losses to a nearby waterbody from an agricultural field having uniform slope, soil type, vegetative cover, and erosion-control practices. It is important to recognize that, for this analysis, the USLE was not modified to calculate the unit soil loss rate; rather, the application of USLE in the overland transport model was modified by introducing the concept of soil mass balance within an integrated drainage subbasin consisting of the LTU, the receiving field (e.g., agricultural field) and surrounding buffer areas. The sediment delivery ratio, defined by empirical data on sediment delivery from drainage basins to receiving streams, was assumed to be constant throughout this integrated setting and used to estimate the fraction of eroded soil (and contaminant) that ultimately reached the field; the soil (and contaminant) that did not reach the waterbody was assumed to be evenly redeposited in the drainage subbasin. Therefore, instead of estimating the soil (and contaminant) load from the LTU to the water body as the USLE soil loss rate times the LTU area, the load was calculated as the USLE soil loss rate times the LTU area times the sediment delivery ratio.

This conceptual approach was first described in Beaulieu et al., 1996, and represented a method to evaluate erosion processes within an integrated setting. Because the USLE equation estimates only soil erosion to waterbodies, the receptor location (i.e., the receiving field) must be located between the LTU and the waterbody. In summary, the equations derived to predict erosion in the integrated setting are based on the following assumptions:

- The area of the management unit (LTU) and the area between the management unit and the nearest waterbody, including the receptor site, make up a discrete drainage subbasin.

- The sediment delivery ratio (SD_{SB}) and the soil loss rate per unit area are assumed to be constant for all areas within the subbasin.

- The amount of soil deposited onto the receptor site through soil erosion is estimated by assuming that the fraction of soil that does not reach the waterbody remains in the subbasin.

- The entire subbasin drainage system is assumed to be at steady-state. Consequently, steady-state soil concentrations for the different subareas (e.g., receptor site, surrounding area) can be calculated using a mass balance approach.

- The soils within the subbasin are assumed (on the average) to have the same soil properties (e.g., bulk density, soil moisture content).

- No contributions to constituent concentrations are assumed to occur from sources other than the LTU within the subbasin.

Background to the Overland Transport Model

The USLE equation contains several factors that influence the amount of soil that is eroded from a site. These factors are all multiplied together to estimate the unit soil loss rate (the amount of sediment that erodes per unit area per unit time) and include:

- rainfall (or erosivity) factor, 1/yr
- erodibility factor, ton/acre
- length-slope factor
- cover management factor
- supporting practice factor.

Researchers have observed that the bulk of eroded sediment is deposited at intermediate locations wherever the entraining runoff waters are insufficient to sustain transport. This effect was quantified as described by Vanoni (1975) using a regression model of the form $SD = aA^b$, where A is the drainage area. The sediment delivery ratio (SD) reflects the empirical observation that the rate of soil erosion observed on small areas (as estimated by the USLE) is diminished as one considers larger and larger areas.

A single sediment delivery ratio was calculated based on the area of the subbasin. That sediment delivery ratio was then assumed to apply to any hypothetically defined subarea within the subbasin, and this was referred to as the "fixed sediment delivery ratio." Although the sediment originating in areas further away from the receiving waterbody would have more opportunity to settle (and, therefore, are less likely to reach the waterbody) than sediment originating in areas directly adjacent to the waterbody, we adopted a simplifying assumption of a fixed sediment delivery ratio for all subareas within the subbasin for two reasons.

First, using the "fixed" sediment delivery ratio for each subarea forced the summation of subarea sediment loads to equal the total sediment load to the receiving waterbody as estimated using the Vanoni correlation for the entire drainage subbasin. Due to the non-linearity of the Vanoni equation, separate application of the equation to the subareas yielded total sediment loads to the waterbody that were different than the sediment load predicted based on the subbasin area. Clearly, the derivation of the empirical relationship presented by Vanoni does not provide the level of resolution necessary to discretize the drainage subbasin into independent drainage subareas. Moreover, the mathematical solution needed to implement such an approach would have required integration of all subareas within the drainage subbasin (e.g., soil movement into and out of each subarea) to ensure that the total soil load to the waterbody matched the summation of the individual loads.

Second, the fixed sediment delivery ratio provides a reasonable estimate of the contaminant load to the receiving waterbody (or stream). This estimate is likely to be somewhat conservative because: 1) the LTU is expected to contribute eroded sediment with the highest contaminant concentration; and 2) the LTU is further away from the receiving waterbody. As the source is the further from the receiving waterbody than the receiving field, soil eroded from the source would have more opportunity to settle in the intervening areas than soil from areas closer to the waterbody (e.g., the field). Therefore, the fixed sediment delivery ratio is expected to provide a reasonably conservative estimate of amount of sediment that erodes directly from the source and reaches the waterbody.

The soil erosion model uses the average soil concentration as calculated for the land treatment unit (LTU) by the source partition model (see Appendix D.1 for detailed description of the source partition model). The overland transport model then assumes that the concentration in the LTU remains constant for the period of time that it takes for the intervening and receiving fields to reach a steady-state concentration. However, the LTU typically has a finite life, and therefore, it is a finite source of contaminated sediment. Time-dependent analysis that were performed under similar modeling exercises suggested that roughly 50 years are needed for the intervening area and receiving field soils to approach steady-state concentrations. Because the contaminant concentration in the LTU will decrease with time (as contaminant mass is eroded), fixing the LTU concentration given the steady-state assumption will tend to produce an overestimate of the contaminant load to both receiving fields and the waterbody. This is a source of uncertainty and a limitation of the current construct.

Site Layout for Modeling the Overland Transport

The Equations shown in Table E-1.5 (as well as the equations in Tables E-1.9 and E-1.29) describe the site geometry of the waste management unit (WMU), field (i.e., receptor site), and receiving waterbody. Implicit in these equations are two simplifying assumptions regarding the relative areas considered in the modeling overland transport. First, the WMU (i.e., the contaminant source) and agricultural field are assumed to be square. Second, the intervening area (defined as the region between the source and the waterbody) is assumed to be at least as large as the source. Therefore, two basic scenarios exist:

1) the source is larger than the receptor field (Figure 1); and
2) the receptor field is larger than the source (Figure 2).

Thus, if the waste management unit (WMU) is larger than the source, the intervening area consists of the field and a buffer that is square and equal in size to the WMU. Conversely, if the WMU is smaller than the field, the field constitutes the intervening area and the up-slope area consists of the WMU and a buffer that is square and equal in size to the field. Given these

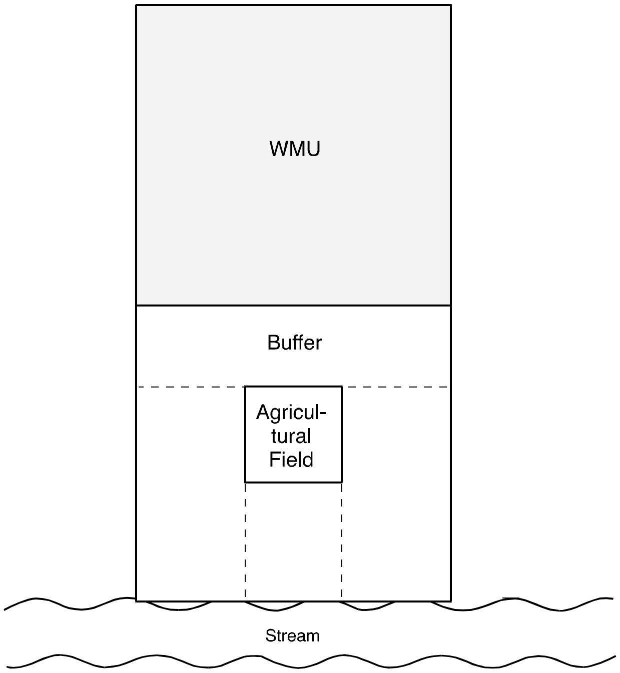

Figure 1. Source > Field

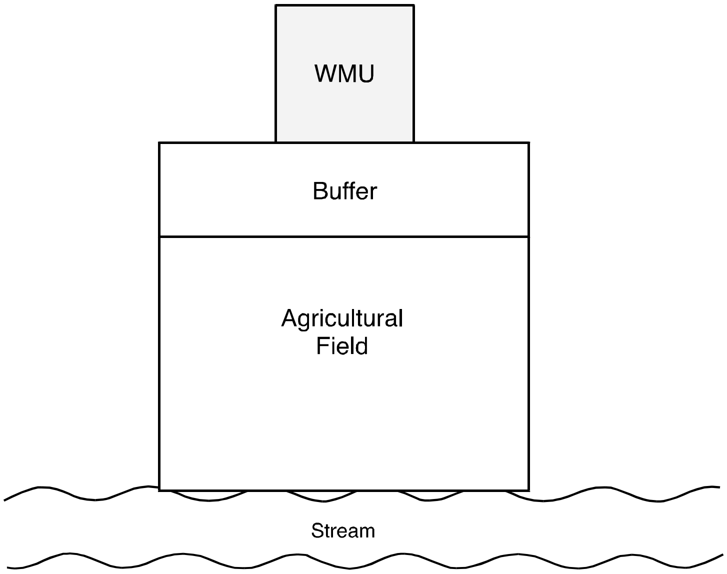

Figure 2. Field > Source.

assumptions illustrated in Figures 1 and 2, the equations derived to estimate for the areas of the buffer and surrounding areas are straightforward, involving simple geometry.

Soil Load from LTU to Receiving Field

The mass of eroded soil (soil load) from the LTU to the receptor site ($SL_{O,F}$) is a major input required to calculate the receptor site soil constituent concentration (C_F). The receptor site (residential plot, home garden, or agricultural field) soil concentrations are used to estimate risk through the soil ingestion pathway for all scenarios and through the food chain pathways (e.g., above-ground and below-ground produce) for the home gardener and farmer scenarios. Assuming that the probability that soil from the LTU is transported and deposited is equivalent for the intervening area and receptor site, the amount of contaminated soil that leaves the source area can be calculated by using a simple ratio of the area of concern to the total area for soil deposition:

$$DS_{0,F} = X_e \times A_s \times (1 - SD_{SB}) \times SF_{0,F} \tag{D-1}$$

where

$DS_{0,F}$ = soil delivery rate from source (LTU) to receptor (kg/yr)
X_e = unit soil loss rate from LTU (kg/m^2/yr)
A_S = area of the LTU (m^2)
SD_{SB} = sediment delivery ratio of the subbasin to the nearest waterbody (unitless)
$SF_{0,F}$ = deposition area scaling factor (m^2/m^2)
= ratio of the receiving field area to the entire area available for deposition
= $A_F/(A_S + A_{B/Surr} + A_F)$

A_F = area of the receiving field (m^2)

$A_{B/Surr}$ = area of the buffer and surrounding areas within the subbasin (m^2).

The sediment delivery ratio (SD) provides an estimate of the fraction of the mobile sediment that directly reaches the waterbody. Consequently, 1- SD provides an estimate of the amount of sediment that does not reach the waterbody and must, therefore, be redeposited in the intervening areas. As suggested previously, the SD reflects the empirical observation that the rate of sediment reaching the stream due to soil erosion observed on small areas (as estimated by the USLE) is diminished as one considers larger and larger areas. Larger drainage basins, which have areas that are further from the receiving waterbody, typically have a lower sediment delivery ratio than smaller drainage basins. Specifically, researchers have observed (from TSS measurements in streams) that “The bulk of the sediment is deposited at intermediate locations wherever the entraining runoff waters are insufficient to sustain transport. It is scattered to adjacent downslope positions; it is deposited at the base of eroding slopes; ...” (Vanoni, 1975). This effect was quantified as described by Vanoni (1975) using a regression model of the form $SD = aA^b$, where A is the drainage area. Using field data from a number of sources that cover a large range of drainage area sizes, researchers estimated the "a" and "b" parameters to be those values such as are used in our model. As seen in Appendix E,Table E-1.4, b=-1/8 and "a" is dependent on the drainage (i.e., subbasin) area.

Because the USLE requires the assumption of homogeneity throughout the basin, and in view of the fact that the SD is not intended to resolve erosion of soil particles by distance from the waterbody, the sediment delivery ratio is assumed to be constant for the drainage subbasin consisting of the LTU, field, and buffer areas. Although the use of the SD does not capture storm events, the model does predict that the majority of the sediment that erodes from the LTU directly reaches the stream, effectively averaging the erosion losses over storm events ranging from mild to severe. In short, the SD reflects annual average sediment transport and was chosen as the most appropriate tool to estimate the long-term sediment yields to the stream based on field measurements. The sediment delivery ratio (SD) is used to estimate how much of the eroding soil reaches the waterbody, and 1-SD is used to estimate how much of that soil is redeposited in the area between the LTU and the receiving stream.

Uniform or even redeposition of sediment is a simplifying assumption used in the model construct; no other assumption appeared technically defensible within the confines of a spreadsheet model. Flow channeling or uneven redeposition could cause all, none, or any fraction of the eroded sediment to be deposited on the receiving field. If one considers that redeposition occurs primarily in flat areas along the run-off flow paths (i.e., uneven redeposition), it still seems likely that the probability of these "areas of significant redeposition" occurring within a given subarea would be proportional to the relative surface areas of the subareas within the subbasin. Thus, although flow channeling would also complicate the steady-state sediment concentrations, the average constituent load to each subarea is reasonably approximated by the uniform (or area dependent) redeposition assumption.

Total Constituent Load to Waterbody

In our conceptual model, the total constituent load to the waterbody (L_T) is the sum of the constituent load from erosion of contaminant-bound soil particles (L_E) and the load from pervious runoff (L_R) from the drainage subbasin. For this analysis, the total constituent load is assumed to originate exclusively from the drainage subbasin, that is, contributions from other subbasins within the watershed are not included. The total load to the waterbody is used to estimate risk to the recreational fisher from the ingestion of contaminated fish. The estimation of L_E requires the calculation of a weighted average constituent concentration in subbasin soils based on the eroded soil contribution ($S_{c,erode}$), and the L_R term requires the calculation of a weighted average constituent concentration based on the pervious runoff[11] contribution ($S_{c,run}$).

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[11] Pervious runoff refers to runoff from watershed soils that are subject to infiltration. Impervious runoff refers to runoff from watershed areas closed to infiltration, such as paved roads. The drainage subbasin was assumed to consist exclusively of pervious soils.

The weighted average constituent concentration represents the effective subbasin soil concentration based on contributions from the three components of the subbasin: the source area (LTU), the receiving field area (garden, agricultural field), and the buffer/surrounding area (intervening area that is not part of the field). The calculation of L_T requires constituent concentrations for each of the following areas within the subbasin: the LTU, the field, and the buffer/surrounding area.

If we consider the constituent load (L_E) to the surface waterbody for each of these areas individually, the equation may be written as:

$$\begin{aligned} L_E = \; & [X_{e,SB} \times ER \times SD_{SB} \times A_0 \times C_0 \times (\frac{Kd_s\ BD}{\theta - Kd_s\ BD}) \times 0.001] - \\ & [X_{e,SB} \times ER \times SD_{SB} \times A_F \times C_F \times (\frac{Kd_s\ BD}{\theta - Kd_s\ BD}) \times 0.001] - \\ & [X_{e,SB} \times ER \times SD_{SB} \times A_{B/Surr} \times C_{B/Surr} \times (\frac{Kd_s\ BD}{\theta - Kd_s\ BD}) \times 0.001]] \end{aligned} \tag{D-2}$$

where

L_E = constituent load to waterbody due to erosion (g/yr)
$X_{e,SB}$ = unit soil loss in subbasin (kg/m^2/yr)
ER = enrichment ratio
SD_{SB} = sediment delivery ratio for subbasin
A_0 = area of source (m^2)
C_0 = constituent concentration at the source (mg/kg)
Kd_s = soil water partition coefficient (L/kg)
BD = bulk density of soil (g/cm^3)
θ = volumetric soil content of soil (cm^3/cm^3)

0.001 = unit conversion factor ([g/kg]/[mg/kg]).

A_F = area of receiving field (m^2)

C_F = constituent concentration in receiving field (mg/kg)

$A_{B/Surr}$ = area of buffer and surrounding area (m^2)

$C_{B/Surr}$ = constituent concentration in buffer and surrounding area (mg/kg)

The enrichment ratio (ER) reflects the tendency for lighter soil particles to erode more easily than heavier particles (lighter soil particles have higher surface-area-to-volume ratios and are higher in organic matter content). The use of an enrichment factor is a well-established practice in watershed modeling, and was considered appropriate for this application as well. Using the ER, the concentrations of organic constituents would be expected to be higher in eroded soil than in situ soil because organics sorb to the soil fines that are most easily eroded. (U.S. EPA 1994). Although light particles tend to erode more easily, our conceptual model for this process does not suggest that light particles are exhausted first, leaving only the larger, heavier particles behind. Indeed, the waste management scenario for the LTU presumes a that the LTU is tilled frequently creating a well-mixed system with respect to grain-size sorting of the soil. Once an erosion event takes place, the fines in surficial soil are preferentially transported as described by the enrichment ratio. The erosion event does not leave behind mostly coarse material; instead, the surficial soil layer becomes once again well-sorted, through mechanical mixing processes such as bioturbation, root action, non-erosion rainfall mixing, etc. so that the ratio of fine to coarse material remains constant across all erosion events as reflected by the enrichment ratio. This simplification of contaminant transport via overland transport of soil is implicit in our approach even though we recognize that the enrichment ratio will not be constant at every point in time; in reality, the enrichment ratio will be a function of the rainfall intensity, soil characteristics, and tilling practices. Nevertheless, the use of an enrichment provides a reasonable method to represent the relationship between soil erosion and long-term average contaminant movement to the waterbody and receiving field.

An alternative way to write Equation D2 can be used to derive an average weighted soil concentration for the entire subbasin (including the LTU, the receptor field, and the buffer/surrounding area) that results in the constituent load as a function of erosion and sediment delivery. The weighted average soil concentration ($S_{c,erode}$) term shown in Equation D-3 is defined as follows:

$$L_E = [X_e \times ER \times SD_{SB} \times A_{SB} \times (\frac{Kd_s\ BD}{\theta - Kd_s\ BD}) \times 0.001] \times S_{c,erode} \qquad \text{(D-3)}$$

where

SD_{SB} = sediment delivery ratio for subbasin
A_{SB} = area of entire subbasin (m^2)

L_T also requires the constituent load from pervious runoff (L_R). The L_R term is calculated using equation D-4.

$$L_R = R \times (A_{SB} - A_I) \times \frac{S_c,erode \times BD}{\theta - Kd_s \times BD} \times 0.01 \qquad \text{(D-4)}$$

where

L_R = pervious surface runoff load (g/yr)
R = average annual surface runoff (cm/yr)
A_{SB} = area of entire subbasin (m^2)
A_I = impervious subbasin area receiving constituent deposition (m^2)

$S_{c,erode}$ = weighted average constituent concentration in total subbasin soils based on surface area (mg/kg)
BD = soil bulk density (g/cm^3)
θ = volumetric soil content of soil (cm^3/cm^3)
Kd_s = soil water partition coefficient (L/kg) or (cm^3/g)
0.01 = units conversion factor ($kg\text{-}cm^2/mg\text{-}m^2$).

Assuming that the ratio of pervious and impervious soils is the same for each of the designated areas, a correction for areas that do not erode (streets, rocks, etc.) can be added to Equation D-3 by replacing A_{SB} with A_{SB} - A_I , where A_I equals the total impervious area in the subbasin. However, our scenario assumes that the entire subbasin consists of pervious soils. Setting the L_R equal in the previous two equations and solving for $S_{c,erode}$ yields:

$$S_{c,erode} = \frac{(X_{e,SB} \times A_s \times C_0 \times SD_{SB}) - (X_{e,SB} \times A_{B/Surr} \times C_{B/Surr} \times SD_{SB}) - (X_{e,SB} \times A_F \times C_F \times SD_{SB})}{X_e \times SD_{SB} \times A_{SB}} \qquad \text{(D-5)}$$

Equation D-5 accounts for differences in the sediment delivery ratios (SD), surface areas (A), and mixing depths (Z) for discrete areas of the subbasin (source, receptor field, and buffer/ surrounding areas). Similarly, the weighted average for runoff losses (ksr) was derived using the areas for various subbasin components. However, as discussed above, different sediment delivery ratios were not required because soils in the area were considered to be similar and the slope was considered uniform. It was possible to generate simple area-based weighting factors because the rainfall runoff per unit area was assumed to be constant for the entire subbasin area.

Constituent Concentrations in Various Subbasin Components

The constituent concentrations for the LTU (C_0), receiving field (C_F), the buffer and surrounding area ($C_{B/Surr}$) are required to solve $S_{c,erode}$. A mass balance approach was used to calculate the constituent concentrations for all subbasin components.[12] For the receiving field, the mass balance equation is given by:

$$M_F\,(dC_F\,/dt) = [(C_0\,SL_{0,F}) - (M_F\,Ds_{(1),F})] - (SL_{B,F}\,C_{B/Surr}) - (M_F\,ks_F\,C_F) \qquad \text{(D-6)}$$

where

M_F = mass of the field (kg)
C_F = constituent concentration in the receiving field (mg/kg)
$SL_{0,F}$ = soil load from source to the field (kg/yr)
$Ds_{(1),F}$ = air deposition rate from source to the field (mg/kg-yr)
$SL_{B,F}$ = soil load from buffer to the field (kg/yr)
ks_F = constituent loss rate coefficient for the field (per yr).

At steady state, this equation can be solved for the constituent concentration in the receiving field as follows:

$$C_F = [(C_0\,SL_{0,F} + M_F\,Ds_{(1),F}) + (SL_{B,F}\,C_{B/Surr})] \,/\, (M_F\,ks_F) \qquad \text{(D-7)}$$

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[12] The mixing depth, along with the surface area, defines the total mass of contaminated soil within a subarea. It defines the mass balance boundaries of the subarea. The contaminant concentration within a subarea is assumed to be homogeneous throughout the subarea (i.e., mixing depth). Contaminant transported below the mixing depth (i.e., via leaching) is effectively lost from the system. The mixing depth is a fundamental parameter in determining the steady state concentrations of the subareas. Different subareas may have different mixing depths depending on the scenario.

As with the constituent concentration in the receptor site field, the concentration in the buffer and surrounding area is given by:

$$\mathrm{M}_{\mathrm{B/Surr}}(d\mathrm{C}_{\mathrm{B/Surr}} / d\mathrm{t}) = (\mathrm{SL}_{0,\mathrm{B/surr}}\ \mathrm{C}_0) + [\mathrm{M}_{\mathrm{B/Surr}}\ (\mathrm{Ds}_{(1),\mathrm{B/Surr}} - \mathrm{ks}_{\mathrm{B/Surr}}\ \mathrm{C}_{\mathrm{B/Surr}})] \qquad \text{(D-8)}$$

where

$\mathrm{M}_{\mathrm{B/Surr}}$ = mass of the buffer and surrounding area (kg)
$\mathrm{C}_{\mathrm{B/Surr}}$ = constituent concentration in the buffer and surrounding area (mg/kg)
$\mathrm{SL}_{0,\mathrm{B/Surr}}$ = soil load from source to buffer/surrounding areas (kg/yr)
C_0 = soil constituent concentration at the source (mg/kg)
$\mathrm{Ds}_{(1),\mathrm{B/Surr}}$ = air deposition rate from source to buffer and surrounding area (mg/kg/yr)
$\mathrm{ks}_{\mathrm{B/Surr}}$ = constituent loss rate coefficient for the buffer/surrounding area (per/yr).

At steady state, this equation may be solved for the constituent concentration in the buffer and surrounding area as follows:

$$C_{B/Surr} = (C_0 SL_{0,B/Surr} - M_{B/Surr}\ Ds_{(1),B/Surr)}\ /\ (M_{B/Surr}\ ks_{B/Surr}) \qquad \text{(D-9)}$$

The constituent loss rate coefficients indicated above (e.g., ks_F) are described in the equation tables provided in Appendix E (see, for example, Equations E-1.15, E-1.16, E-1.19, E-1.21, E-1.22). For each subarea in the intervening field, contaminant losses due to leaching, run-off, erosion, degradation and volatilization are estimated assuming first order decay rates. The leaching and run-off rates assume that the leaching and run-off liquid reach equilibrium with the soil at the specified subarea. Erosion rate losses are calculated as part of the overland transport model, and are consistent with the assumptions and calculations used for the model (in terms of

USLE, sediment delivery and area-based scaling factors). Degradation rate constants are based on contaminant half-lives in soils as reported by Howard (1991); these are predominately biodegradation rates, but may also include hydrolysis rates. Volatilization rate constants are estimated using the gas phase mass transfer coefficient correlation of MacKay and Matasugu as reported in Hwang (1982). Each of the individual loss rate coefficients are summed to calculate the overall first order contaminant disappearance rate for the given subarea; this overall constituent loss rate coefficient reduces the steady state concentration in the various subareas, including the field receptor sites. Contaminant loads to a subarea are limited to soil erosion and air deposition. Run-off from the source is assumed to transport contaminants directly to the receiving waterbody. Run-off from the source does not infiltrate the off-site field, nor does it contribute to the contaminant load for the off-site field.

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Appendix D.4

Groundwater Contaminant Fate and Transport Modeling Using EPACMTP

Appendix D.4

Groundwater Contaminant Fate and Transport Modeling Using EPACMTP

Background Information on EPACMTP

EPACMTP (EPA's Composite Model for Leachate Migration with Transformation Products) (U.S. EPA, 1996a, 1996b, 1996c, and 1997a) is a computer simulation model for modeling the subsurface fate and transport of contaminants leaching from a land disposal site, e.g., landfill, surface impoundment, wastepile, or land application unit. Fate and transport processes accounted for in the model are: advection, hydrodynamic dispersion, equilibrium linear or nonlinear sorption, and chemical and biological decay processes. The composite model consists of two coupled modules: (1) a one-dimensional module that simulates infiltration and dissolved contaminant transport through the unsaturated zone, and (2) a saturated zone flow and transport module that can be run in either 3-D or quasi 3-D mode. EPACMTP also has the capability to perform Monte Carlo simulations to account for parametric uncertainty or variability. The flow and transport simulation modules of EPACMTP are linked to a Monte Carlo driver, which permits a probabilistic evaluation of uncertainty in model input parameters, as described by specified (joint) probability distributions.

EPACMTP has been published in an international refereed journal (Kool et al., 1994) and has been reviewed by EPA's Science Advisory Board (U.S. EPA, 1995). This review commends the Agency for its significant improvements to the model and states that EPACMTP represents the state of the art for nationwide regulatory analyses (U.S. EPA, 1995).

Methods and Assumptions Used to Model Flow and Transport

EPACMTP simulates steady-state flow in both the unsaturated zone and the saturated zone; contaminant transport can be either steady state or transient. The steady-state modeling option is used for continuous source modeling scenarios; the transient modeling option is used for finite source modeling scenarios. The output of EPACMTP is a prediction of the contaminant concentration arriving at a downgradient groundwater receptor well. This can be either a steady-state concentration value, corresponding to the continuous source scenario, or a time-dependent concentration, corresponding to the finite source scenario. In the latter case, the model can calculate either the peak concentration arriving at the well, or a time averaged concentration, corresponding to a specified exposure duration, e.g., a 9-year average residence time.

Flow in the Unsaturated Zone. Flow in the unsaturated zone is assumed to be steady-state, one-dimensional vertical flow from beneath the source toward the water table. The lower boundary of the unsaturated zone is assumed to be the water table. Actual flow in the unsaturated zone is predominantly gravity-driven. Therefore, it is reasonable to model flow in the unsaturated zone as one-dimensional in the vertical direction. It is also assumed that transverse dispersion (both mechanical dispersion and molecular diffusion) is negligible in the vadose zone. This assumption is based on the fact that lateral migration due to transverse dispersion is negligibly small compared with the horizontal dimensions of waste management units. In addition, this assumption is conservative because it allows the leading front of chemicals to arrive at the water table relatively sooner and, in the case of a finite source, with greater peak concentration. The flow rate is assumed to be determined by the long-term average infiltration rate through the waste management unit (WMU).

Transport in the Unsaturated Zone. Contaminant transport in the unsaturated zone is assumed to occur by advection and dispersion. The unsaturated zone is assumed to be initially contaminant-free, and contaminants are assumed to migrate vertically downward from the disposal unit. EPACMTP can simulate both steady-state and transient transport in the unsaturated zone with single-species or multiple-species chain decay reactions and with linear or nonlinear sorption.

Flow in the Saturated Zone. The saturated zone module of EPACMTP is designed to simulate flow in an unconfined aquifer with constant saturated thickness. The model assumes regional flow in a horizontal direction with vertical disturbance resulting from recharge and infiltration from the overlying unsaturated zone and waste disposal unit, respectively. The lower boundary of the aquifer is assumed to be impermeable. Flow in the saturated zone is assumed to be steady-state. EPACMTP accounts for different recharge rates beneath and outside the source area. Ground water mounding beneath the source is represented in the flow system by increased head values at the top of the aquifer. This approach is reasonable as long as the height of the mound is small relative to the thickness of the saturated zone.

Transport in the Saturated Zone. Contaminant transport in the saturated zone is assumed to be the result of advection and dispersion. The aquifer is assumed to be initially contaminant-free, and contaminants are assumed to enter the aquifer only from the unsaturated zone immediately underneath the waste disposal unit, which is modeled as a rectangular, horizontal plane source. EPACMTP can simulate both steady-state and transient three-dimensional transport in the aquifer. For steady-state transport, the contaminant mass flux entering at the water table must be constant with time; for the transient case, the flux at the water table may be constant or may vary as a function of time.

Treatment of Chemical-Specific First-Order Decay and Sorption

EPACMTP can simulate the transport of single-species or multiple-species chain decay reactions and also accounts for chemical and biological transformation processes. All transformation reactions are represented by first-order decay processes. These transformation processes can be lumped together and specified as an overall decay rate or specified with separate first-order decay coefficients for chemical decay and biodegradation. EPACMTP also has the capability to determine the overall decay rate from chemical-specific hydrolysis constants using soil and aquifer temperature and pH values. In the event that the daughter products of transformation are hazardous and their chemical-specific parameters are known, the model can also account for the formation and subsequent fate and transport of these daughter products.

The groundwater pathway analysis accounts for equilibrium sorption of waste constituents by the soil and aquifer solid phase. For organic constituents, a partition coefficient (K_d) is calculated as the product of the constituent-specific organic carbon partition coefficient (K_{oc}) and the fraction organic carbon (f_{oc}) in the soil and aquifer.

The metals modeling methodology in EPACMTP incorporates two options to specify the K_d for a given metal. Adsorption isotherms for metals with nonlinear sorption behavior are computed using EPA's geochemical speciation model, MINTEQA2 (Allison et al., 1991); and the isotherms for metals which cannot be accurately modeled with MINTEQA2 are specified as pH dependent empirical relationships (Loux et al., 1990). The two approaches for calculating the K_d values are described briefly below; more detailed information can be found in U.S. EPA (1996c and 1997a).

In the first approach, the purpose of using the MINTEQA2 model is to capture the variation in K_d due to variability in geochemical conditions in the soil and changing dissolved metal concentrations. The four geochemical parameters on which adsorption is assumed to primarily depend are: groundwater pH, concentration of hydrous ferric oxide adsorption sites, concentration of dissolved and particulate natural organic matter, and concentration of leachate organic acids. For the MINTEQA2 modeling, the natural variability of these parameters is divided into three ranges: low, medium, and high. Then, each parameter was assigned three possible values, which correspond approximately to the midpoint of each range. For each metal with nonlinear adsorption, the MINTEQA2 model was then run over a range of total metal concentrations to produce an isotherm for each combination of the three possible values for the four geochemical parameters. For each metal, the 162 isotherms produced in this way were then written to a data file that must accompany the input file when conducting EPACMTP modeling. EPACMTP then selects the appropriate isotherm based on the input values specified for the four geochemical parameters.

To perform geochemical modeling with MINTEQA2, one must know the adsorption reactions describing the interaction of the metal with the adsorbing surface. For several metals of concern, primarily those that behave as anions in aqueous solution, these reactions are not reliably known. Because the MINTEQA2 model could not be used due to this lack of data, a second approach was developed that uses empirical linear relationships to describe the adsorption distribution coefficient as a function of pH. The pH-dependent isotherms were determined from statistical analysis of laboratory measurements of soil and aquifer materials and corresponding groundwater and leachate samples (Loux et al., 1990). These isotherms are included in the EPACMTP code, and the appropriate K_d value is calculated based on the input value specified for groundwater pH in the unsaturated and saturated zones.

Deterministic Modeling For The Chlorinated Aliphatics Listing Determination

The risk analysis for the chlorinated aliphatics listing determination consists of three phases of modeling for the landfill and land treatment unit scenarios: a deterministic sensitivity analysis, a deterministic 2-parameter high end analysis, and a probabilistic analysis. The first two phases will be described in this section.

The sensitivity analysis is used to identify and rank the most influential variables in the analysis. For the chlorinated aliphatics listing determination, the sensitivity analysis was conducted in deterministic mode; that is, each input is set to a constant value, one model realization is performed, and the model outputs one receptor well concentration. Additionally, this sensitivity analysis separately evaluated the influence of each of a number of input parameters. The first step of the sensitivity analysis was to run EPACMTP with all input parameters set to their respective central tendency (median) values. Then, one at a time, each input to be examined in the sensitivity analysis was set to its high end (usually the 10th or the 90th percentile) value and the variation in the receptor well concentration from the central tendency case is noted. The inputs having the greatest impact on the receptor well concentration are identified as the most sensitive parameters. Note that the highest 9 or 30-year average receptor well concentration is used for carcinogenic constituents and the highest 9-year average receptor well concentration is used for non-carcinogenic constituents.

The two most sensitive parameters identified in the sensitivity analysis are then set to their respective high end values in the high end analysis, while the remaining parameters are set to their median values. The resulting receptor well concentration is defined as the high end concentration.

Deterministic Sensitivity Analysis

The purpose of the groundwater pathway sensitivity analysis is to identify the most sensitive parameters in the exposure and risk calculations, and their corresponding high end and central tendency values for the subsequent deterministic analysis. The sensitivity of individual parameters can be expressed as the difference, or ratio, in predicted health risk or hazard when the parameter is set to its high end value, compared to the risk corresponding to the central tendency value of that parameter. The high end value of a parameter corresponds to its 90th percentile value or its 10th percentile value, depending on whether a high or a low value of that parameter results in a higher predicted risk. If there is limited data to define the probability distribution of a parameter, the high end may be set to either the maximum or minimum measured value. The central tendency value corresponds to the 50th percentile (median) value of the parameter.

Identification and Description of Important Parameters

The various parameters can be grouped into constituent-related parameters, waste- and WMU-related parameters, pathway-related parameters, and intake-related parameters. This discussion does not include all parameters in the groundwater model or in the exposure and risk equations, but is restricted to those that are expected to be among the most sensitive parameters.

Constituent-related Parameters. The most important parameters in this group are:

- Concentration of constituent in the waste
- Concentration of constituent in the leachate
- Organic carbon partition coefficient (for organics), or the solid-liquid partition coefficient (for metals and inorganics)
- Transformation (hydrolysis) half-life

For the landfill scenario, the constituent-specific leachate concentration was included in the sensitivity analysis and was based on industry-specific sampling data. For dioxins, the Toxicity Characteristic Leaching Procedure (TCLP) concentration for sample OG-04 was used as the central tendency value, and the TCLP concentration for sample GL-01 was used as the high end value for leachate concentration. For all other constituents, the central tendency value for leachate concentration was calculated as the average TCLP concentration and the high end value was defined as the maximum TCLP concentration.

However, the constituent-specific waste concentration was not included in the sensitivity analysis for landfills because the TCLP data did not necessarily correspond to the mean and max waste concentration. Note that waste concentration data have been compiled based on dry weights and wet weights; the wet weight concentrations are lower than the dry weight concentrations, and are the appropriate concentrations for use in our analyses since the corresponding volumes of the wastes in question were estimated on a wet basis, and the wastes are not dried prior to disposal (they are disposed "wet"). The effect of varying waste concentration is similar to that of waste volume (which was included in the sensitivity analysis); increasing either of these parameters will increase the leaching duration, and drive the exposure concentration at the receptor well toward a steady state value. For the dioxins, the dry weight waste concentration for sample OG-04 was used as the central tendency value, and the dry weight waste concentration for sample GL-01 was used as the high end value for waste concentration. For bis(2-chloroethyl)ether and arsenic the average wet weight values from industry-specific sampling data were used for waste concentration. For all other constituents, the constituent-specific value for waste concentration was calculated as the average dry weight waste concentration. Since the dry weight concentrations are higher than the wet weight concentrations, and since no constituents for which the analysis was performed based on dry weight concentrations showed excessive risk, these analyses were not repeated using the correct wet weight concentrations.

For the land treatment unit scenario, leachate concentration is an input for the groundwater model, but waste concentration is not. The groundwater model uses the conservative assumption that during operation of the LTU, the source is periodically replenished through additional waste applications, so that no significant depletion of the source occurs. In other words, the leachate concentration emanating from the LTU remains constant. The magnitude of the leachate concentration in this scenario is controlled by the waste concentration, as determined by the source partitioning model. The effect of waste concentration was, therefore, included in the modeling analysis, but this parameter was not used as a separate model input parameter. Central tendency and high end leachate concentration values were modeled by varying the waste concentration in the partitioning model from its central tendency value to its high end value, respectively.

The last two constituent-specific parameters, partition coefficient (k_d) or organic carbon partition coefficient (k_{oc}), and the hydrolysis half-life, are considered to be properties of the constituents which do not generally vary. Thus, these parameters were not considered in the sensitivity analysis.

Waste and Waste Management Unit-Related Parameters. Landfills and land treatment units are the two types of waste management units (WMUs) that were evaluated in this risk assessment. The significant parameters in this category include:

- WMU surface area
- Infiltration rate through the WMU
- WMU operating life
- Annual waste amount

The product of WMU area and infiltration rate equals the annual volumetric leachate flux through the WMU. The product of leachate flux times leachate concentration equals the annual mass of constituent that is released into the subsurface. For a given WMU area and leachate concentration, a higher infiltration rate means a higher loading of contaminant into the soil and groundwater, but, for the landfill scenario, also a more rapid depletion of the constituent. Assuming a uniform WMU design (earthen cover, no liner), the infiltration rate is controlled by climatic factors, i.e., it will vary depending on the geographic location of the waste management unit.

The landfill operating life and the annual quantity of waste disposed determine the total amount of waste that accumulates in the landfill. Landfill operating life was not varied in the sensitivity analysis because 30 years has been defined as the average operating life for municipal landfills (U.S. EPA, 1997b).

For the land treatment unit scenario, the constituent is assumed to leach into the subsurface at a constant rate and at a constant concentration during the active life of the unit. After the unit no longer accepts sludge for land application, it is assumed that the amount of leachate generated will quickly diminish. Thus, it was conservatively assumed that after cessation of sludge application, the generation of leachate would continue for no more than 40 years. Adding these 40 years of leaching after unit closure to the assumed 40 years of active use of the land application unit produces the leaching duration of 80 years for this waste disposal scenario. Thus, land application unit active life and leaching duration were not varied in the sensitivity analysis.

The annual waste quantity is characterized by considerable uncertainty and variation; therefore, this parameter was examined in the sensitivity analysis.

Groundwater Pathway Related Parameters. The most important parameters affecting dilution and attenuation in the soil and groundwater include:

- Soil type and soil characteristics (including saturated conductivity and water content)
- Depth to groundwater
- Saturated zone thickness
- Aquifer hydraulic conductivity

- Hydraulic gradient
- Distance to nearest receptor well
- Depth of well intake point
- Position of well relative to plume centerline

In support of the 1995 HWIR proposal, a methodology and database were developed to relate a number of the most important soil and groundwater parameters to waste unit location (U.S. EPA 1997a and 1997b). These location-dependent parameters are: (1) depth to groundwater, (2) saturated zone thickness, (3) aquifer hydraulic conductivity, and (4) hydraulic gradient. These data were used for this modeling analysis to determine the value of each of these parameters at each facility location modeled.

Whereas distance to nearest receptor well and position of the well relative to plume centerline were examined in the sensitivity analysis, depth of well intake point was not included. There can be significant differences in groundwater concentration at different depths below the water table. However, the depth at which the maximum concentration occurs varies according to well location and the hydrogeologic setting being modeled. Therefore, for the sensitivity analysis, the well was placed at the vertical mid-point of the aquifer.

It is important to note that the dilution and attenuation of waste constituents during transport in the groundwater pathway depends strongly on the pathway-related parameters identified above and on the constituent-specific sorption (k_d or k_{oc}) parameters and hydrolysis transformation rate. The effect of sorption (high k_{oc} or k_d) is to retard the movement of constituents relative to the rate of groundwater movement, thereby increasing the travel time through both the unsaturated zone (from the base of the landfill to the water table) and the saturated zone to the receptor well.

For constituents that do not hydrolyze, the primary effect of this retardation is to delay the time of maximum exposure. For relatively large waste volumes (in which contaminant transport approaches steady state conditions), the magnitude of the exposure at the receptor well is less affected. For smaller waste volumes, the magnitude of the exposure at the receptor well may be significantly affected by retardation. For constituents that do hydrolyze, an increased travel time means that a greater proportion of the constituent mass will have transformed before it reaches the receptor well, which may result in lower exposure and risk (although the risk associated with toxic transformation daughter products may be increased). Thus, the relative sensitivity of "depth to groundwater" and "distance to receptor well" may be markedly different for different constituents.

Intake Related Parameters. Parameters in this category include:

- Exposure duration (for carcinogens only)
- Exposure frequency
- Groundwater intake (ingestion) rate
- Body weight

The sensitivity analysis includes evaluation of both modeling parameters and intake-related parameters. Since the risk equation is linear, these intake parameters can be evaluated directly, without the use of groundwater modeling. For instance, if all other factors remain constant, a doubling of the ingestion rate doubles the risk. Moreover, the parameters in this group generally act independently of any of the other parameters discussed before (all of which affect the exposure concentration), and are generally independent of the constituent being analyzed.

Exposure duration, applicable for carcinogens only, is the exception to these generalizations. Exposure duration is the most sensitive of the intake-related parameters; its central tendency value is 9 years, and its high end value is 30 years for adult residents (U.S. EPA, 1997c). In order to maintain consistency between the modeled groundwater transport scenario and the risk analysis scenario, exposure duration was examined in the sensitivity analysis for carcinogenic constituents. Given a constant receptor well concentration, increasing the exposure duration (from 9 years to 30 years) for carcinogens increases risk by slightly more than a factor of 3 (the ratio of 30 to 9).

Since the ratio of high end receptor well concentration to central tendency well concentration for the first most sensitive parameter is greater than three, the ratio for the second most sensitive parameter was calculated and compared to that of exposure duration. If the ratio for the second most sensitive parameter is greater than three, then it was chosen as the second parameter for the high end analysis and the central tendency value of 9 years was used for exposure duration. Conversely, if the ratio for the second most sensitive parameter is less than 3, then exposure duration (30 years) was chosen as the second parameter for the high end analysis. For non-carcinogens, the 9-year average receptor well concentration always was used.

Summary of Simulation Procedure

The sensitivity of individual parameters is defined as the difference, or ratio, in predicted health risk or hazard when the parameter is set to its high end value, compared to the risk corresponding to the central tendency value of that parameter. The high end value of a parameter corresponds to its 90th percentile value or its 10th percentile value, depending on whether a high or a low value of that parameter results in a more conservative (higher) predicted risk. If there is limited data to define the probability distribution of a parameter, the high end may be set to either the maximum or minimum measured value. The central tendency value corresponds to the 50th percentile (median) value of the parameter.

The sensitivity analysis was conducted by performing a number of modeling runs for each constituent. First, all parameters were set to their central tendency values. Then one at a time, each parameter was set to its high end value while all the other parameters remained at their central tendency values. These values and the data sources for the landfill scenario and the land treatment unit scenario are presented in Appendix K. The list of parameters examined in the sensitivity analysis for landfills and land treatment units are presented in Tables D.4-1 and D.4-2. The modeling results were then tabulated, and the parameters, including intake parameters, were ranked in order of sensitivity for each group of constituents. Finally, the two most sensitive parameters were identified for use in the subsequent deterministic analysis. The complete groundwater modeling results for the sensitivity analysis are presented in Appendix H. These results are summarized here in Tables D.4-3 and D.4-4. Note that cis-1,3-dichloropropene and dioxins did not reach the receptor well. The neutral hydrolysis rate constant and K_{oc} used to model cis-1,3-dichloropropene were 40 yr^{-1} and 63.1 cm^3/g, respectively (Kollig 1993). Data for dioxins are presented in Appendix C.

Table D.4-1. Parameters Examined in the Sensitivity Analysis for the Landfill Scenario

Landfill Parameters
X-well
Y-well
Area
Site Location
Leachate Concentration
Waste Volume
Exposure Duration (carcinogens only)

Table D.4-2. Parameters Examined in the Sensitivity Analysis for the Land Treatment Unit Scenario

Land Treatment Unit Parameters
Leachate Concentration
X-well
Y-well
Exposure Duration (carcinogens only)

Table D.4-3. Results of Sensitivity Analysis for Landfill Scenario

Constituent of Concern	Two Most Sensitive Parameters
1,2-Dichloroethane	X-well and Area
Chloroform	X-well and Area
Methylene chloride	X-well and Exposure Duration
cis-1,3-Dichloropropene	Not determined [a]
OCDD	Not determined [a]
OCDF	Not determined [a]
1,2,3,4,7,8,9-HpCDF	Not determined [a]
1,2,3,4,6,7,8-HpCDF	Not determined [a]
Bis(2-chloroethyl)ether	Site Location and X-well
2-(2-chloroethoxy)ethanol	Site Location and X-well
1,4-Dioxane	X-well and Area
Arsenic	X-well and Exposure Duration
Manganese	X-well and Area
Molybdenum	X-well and Waste Volume
Nickel	Site Location and X-well

[a] The relative sensitivity of the parameters could not be determined because the plume did not reach the receptor well for the central tendency scenario or any of the high end scenarios.

Table D.4-4. Results of Sensitivity Analysis for Land Treatment Unit Scenario

Constituent of Concern	Two Most Sensitive Parameters
Arsenic	Leachate Concentration and Exposure Duration
Bis(2-chloroethyl)ether	X-well and Exposure Duration
2-(2-chloroethoxy)ethanol	X-well and Exposure Duration
1,4-Dioxane	X-well and Exposure Duration

Deterministic Analysis

The purpose of the groundwater pathway deterministic analysis for the chlorinated aliphatics waste streams is to predict the potential chemical-specific high end and central tendency risks for the landfill and land treatment unit scenarios. The high end scenario is defined by setting the two most sensitive parameters to their respective high end values and setting all other parameters to their central tendency values. In addition, a deterministic central tendency analysis, in which all parameters are set to their central tendency values, was performed.

Linkage of Source Partitioning Model and EPACMTP

The source model developed by Research Triangle Institute (RTI) was used to simulate the major fate and transport processes, such as leaching, hydrolysis, and volatilization, that occur within the waste management unit. The partitioning model and the EPACMTP model can be run sequentially, as long as all parameters that are common to both models, e.g. landfill area and infiltration rate, are set to the same values in both models. If this is not done, then conservation of contaminant mass is not maintained.

For the landfill scenario, the partitioning model was used to predict the reduction in constituent amount due to volatilization during the active life of the unit. The waste concentration used for the groundwater pathway modeling for the landfill scenario was adjusted to account for these volatilization losses. The TCLP concentration was used as the initial leachate concentration for the landfill scenario. Appendix D.1 provides a complete description of the partitioning modeling that was performed to simulate the leaching process for the landfill waste management scenario.

For the land treatment unit scenario, the source partitioning model was used to generate a leachate profile for each constituent to be modeled; that is, a history of the annual average leachate concentrations. The maximum 9-year average leachate concentration during the 80 years after the opening of the unit was used as the leachate concentration for the land treatment unit scenario. Appendix D.1 provides a complete description of the partitioning modeling that was performed to simulate the leaching process for the land treatment unit scenario.

Results of the Deterministic Analysis

Some constituents did not reach the receptor well within the 10,000 year modeling period for any of the high end scenarios; thus, for these constituents, a two-parameter high end analysis was not conducted. These constituents are cis-1,3-Dichloropropene, OCDD, OCDF, 1,2,3,4,7,8,9-HpCDF, and 1,2,3,4,6,7,8-HpCDF and are generally characterized by very high K_{OC} values. The EPACMTP model would predict that they are only very slowly mobile in the subsurface. The results of the deterministic central tendency and high end analysis for the landfill and land treatment unit scenarios for the remaining constituents are presented below in Tables D.4-5 and D.4-6.

Table D.4-5. Results of the Deterministic Analysis for the Landfill Scenario

Constituent of Concern	Scenario	Receptor Well Concentration (mg/L)
1,2-Dichloroethane	Central Tendency	8.85E-05
	High end X-well and Area	1.26E-03
Chloroform	Central Tendency	1.16E-04
	High end X-well and Area	1.03E-03
Methylene chloride	Central Tendency	1.43E-04
	High end X-well and Exposure Duration	8.98E-04
Bis(2-chloroethyl)ether	Central Tendency	1.19E-07
	High end Site Location and X-well	1.30E-04
2-(2-chloroethoxy)ethanol	Central Tendency	2.82E-07
	High end Site Location and X-well	7.29E-04
1,4-Dioxane	Central Tendency	1.12E-04
	High end X-well and Area	1.46E-03
Arsenic	Central Tendency	2.10E-04
	High end X-well and Exposure Duration	1.42E-03
Manganese	Central Tendency	1.18E-01
	High end X-well and Area	1.55E+00
Molybdenum	Central Tendency	2.48E-04
	High end X-well and Waste Volume	4.75E-03
Nickel	Central Tendency	1.62E-06
	High end Site Location and X-well	3.08E-01

Table D.4-6. Results of the Deterministic Analysis for the Land Treatment Unit Scenario

Constituent of Concern	Scenario	Receptor Well Concentration (mg/L)
Arsenic	Central Tendency	1.88E-04
	High end Leachate Concentration and Exposure Duration	5.00E-04
Bis(2-chloroethyl)ether	Central Tendency	4.45E-06
	High end X-well and Exposure Duration	1.16E-05
2-(2-chloroethoxy)ethanol	Central Tendency	1.08E-05
	High end X-well and Exposure Duration	2.39E-05
1,4-Dioxane	Central Tendency	2.84E-05
	High end X-well and Exposure Duration	4.73E-05

Probabilistic Modeling For The Chlorinated Aliphatics Listing Determination

The purpose of the groundwater pathway Monte Carlo analysis is to provide a probabilistic estimate of risk given the uncertainty and variability in groundwater pathway parameters. The Monte Carlo analysis also provides an estimate of where in the distribution of risk the results of the deterministic analysis fall. For this analysis, the output of the groundwater fate and transport model is a probability distribution of receptor well concentrations. The Monte Carlo analysis was only conducted for constituents which showed significant (for example, greater than $1x10^{-6}$) risk in the high end deterministic analysis. For both the landfill and land treatment unit scenarios, arsenic was the only constituent for which the Monte Carlo modeling was performed.

Appendix F.1 provides a complete description of the EPACMTP site-based methodology, the data sources for probability distributions of input parameters, and a detailed discussion of how the source partitioning model and EPACMTP were linked in the Monte Carlo analysis. The groundwater concentrations predicted in the Monte Carlo analysis are presented in Appendix H.

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Appendix D.5

Surface Water Model

Appendix D.5

Surface Water Model

Introduction

The framework for estimating surface water impacts from the management of chlorinated aliphatics wastes is based on the methodology presented in *Addendum to Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions* (U.S. EPA, 1993), henceforth referred to as the Addendum. The model estimates water column and bed sediment concentrations. Fish tissue concentrations are estimated from water column or sediment concentrations using bioconcentration factors (BCFs), bioaccumulation factors (BAFs), or biota-sediment accumulation factors (BSAFs).

Water column concentrations include dissolved, sorbed to suspended sediments, and total concentration (sorbed plus dissolved). The model accounts for five routes of contaminant entry into the waterbody: (1) contaminant sorbed to eroding soils, (2) contaminant dissolved in runoff, (3) particle-bound contaminant deposition, (4) wet deposition of vapor phase contaminant, and (5) direct diffusion of vapor phase contaminants. In addition, the model will accept input from the groundwater model on contaminated groundwater that enters the waterbody through the subsurface. The model also accounts for processes that remove contaminants. These include: volatilization of dissolved phase contaminants from the water column and contaminant removal via burial. The impact to the waterbody is assumed to be uniform, which is more realistic for smaller waterbodies than for larger ones. Key assumptions in the model include the following:

- Soil concentrations within the depositional area are uniform
- Concentrations within the surface soils, sediments, suspended solids, and water can be described by partition coefficients
- At equilibrium, gaseous diffusion into the water equals volatilization from the water
- Equilibrium is maintained between contaminants within the water column and contaminants in sediment
- Rate of contaminant burial in bed sediments is estimated as a function of the sediment deposition rate.

The scenario modeled for to predict fisher risks simulates contaminant loadings to a reach order 3 stream located at either a high end (102 m) or central tendency distance (430 m for the LTU; 300 m for other units) from the source to the stream.

Waterbody/Watershed Characterization

The parameters contained in this section characterize the surface waterbody (a stream) simulated by the model. The waterbody characterization parameters are another example of a set of parameters that are interdependent; therefore, they are set and varied as a group.

Van der Leeden et al. (1990) ranked over 2 million streams located throughout the United States according to their stream order. A first-order stream has no tributary channels; a second-order stream forms when two first-order streams converge, and so on through stream order 10. Each successive stream order is characterized by a larger flow volume. For each stream order, van der Leeden presented typical values for flow, waterbody area, watershed area, depth, and various other parameters. A third order stream reach was chosen as the waterbody for the modeling simulations in this analysis and was characterized using van der Leeden's typical values for the third order stream reach. Van der Leeden's data on stream reaches are reproduced in Table D.5-1.

Stream orders 1 and 2 are typically too small to support the fish consumption assumed under the fisher scenario and, therefore, were eliminated from consideration. Of the stream orders sufficiently large to support aquatic life for fishing (stream orders 3 through 10), we selected stream order 3 to create an exposure scenario that was protective of different types of fishable waterbodies and, at the same time, representative of streams on a national basis. As shown in Table D.5.1, reach order 3 streams are more prevalent in terms of sheer number than all other fishable streams and rivers combined; thus, the reach order 3 streams would be expected to be accessible to fishers in most areas of the country. In addition, the total length for reach order 3 streams is within 10% of all the other fishable streams and rivers combined. Lastly, the conceptual framework for this simulation uses a single point estimate for air concentration and deposition rate. As an approximation, use of a single value is reasonable for a small stream such as that described under reach order 3. As noted in the Technical Background Document, this scenario was developed to be protective of all recreational and subsistence fisher scenarios. However, using a single point estimate for a reach order 5 stream was not considered to be a reasonable approach to estimating fish concentrations, and would have resulted in a significant overestimate of the fisher risks. Based on this information, a reach order 3 stream was used as the fishable waterbody in the risk modeling simulations for the fisher scenario.

Table D.5-2 summarizes the stream data used to characterize the reach order 3 stream. These values are converted to metric units, as needed by the model. Watershed area (called drainage area by van der Leeden), flow, depth, and velocity were taken directly from van der Leeden. Waterbody area was calculated from average length and width. Flow-independent mixing volume was calculated from average length, width, and depth, as suggested in the Addendum.

Table D.5-1. Summary of U.S. Stream Data

Stream order	Number of streams	Total length (mi)	Average length (mi)	Drainage area (mi^2)	Surface area (mi^2)	Mean flow (ft^3/s)	Mean width (ft)	Mean depth (ft)	Mean velocity (ft/s)
1	1,570,000	1,570,000	1.0	1.0	1,200	0.65	4	0.15	1.0
2	350,000	810,000	2.1	4.7	1,500	3.1	10	0.29	1.3
3	80,000	420,000	5.3	23	1,400	15	18	0.58	1.5
4	18,000	220,000	12	109	1,500	71	37	1.1	1.8
5	4,200	116,000	28	518	1,600	340	75	2.2	2.3
6	950	61,000	64	2,500	1,800	1,600	160	4.1	2.7
7	200	30,000	147	12,000	1,800	7,600	320	8.0	3.3
8	41	14,000	338	56,000	1,700	36,000	650	15	3.9
9	8	6,200	777	260,000	1,500	171,000	1,300	29	5.6
10	1	1,800	1,800	1,250,000	1,000	810,000	2,800	55	5.9

Source: Van der Leeden et al. (1990).

Table D.5-2. Characterization of Reach Order 3 Stream

Parameter	**Stream Order 3**
Watershed area	6e+7 m^2
Flow	1.3e+7 m^3/yr (1.3e+10 L/yr)
Velocity	0.5 m/s
Depth (waterbody)	0.18 m
Width	5.5 m
Length	8,500 m
Waterbody area (length × width)	4.6e+4 m^2
Flow-independent mixing volume (length × width × depth)	8.3e+3 m^3 (8.3e+6 L)
Depth (bed sediment)	0.03 m
Depth (water column)	0.15 m

The surface water model requires three different depth measurements as inputs: depth of the water column, depth of bed sediment, and total waterbody depth (which is the sum of the water column and sediment depths). The depth from van der Leeden was for total waterbody depth. The Addendum suggests a typical bed sediment depth of 0.03 m; this was used, and the water column depth calculated as the difference between the total waterbody depth from van der Leeden and the bed sediment depth of 0.03 m.

Other Surface Water Parameters

Total Suspended Solids

The Addendum suggests that total suspended solids (TSS) can range from 1 to 100 mg/L and suggests a “typical” value of 10 mg/L for streams and rivers. However, given the depth of the stream and flow rate, we might expect to have a higher value for TSS than the “typical” value recommended for streams and rivers of all sizes (e.g., small, shallow streams will have less settling of suspended solids because of turbulence). The Addendum suggests that 80 mg/L is a cutoff value for protection of aquatic life and, therefore, this value was selected for the reach order 3 stream. It should be noted that varying the TSS concentration has little impact on the modeling results on metals and hydrophilic organic constituents (i.e., organic chemicals with the log K_{ow} value below 4.0). However, for highly hydrophobic constituents such as dioxins, decreasing the TSS to 10 mg/L from 80 mg/L results in less than a two-fold increase in risks from ingestion fish, with all other parameters set to the same value. Thus, the model is only moderately sensitive to this parameter, and the default TSS value of 80 mg/L is considered a minor source of uncertainty.

Bed Sediment Concentration

The bed sediment concentration term is analogous to the bulk density for soil in that it describes the concentration of solids in terms of a mass per unit volume. The Addendum notes that bed sediment concentration should range from 0.5 to 1.5 kg/L and that a reasonable value for most applications is 1 kg/L. Given the narrow range for this parameter, the recommendation in the Addendum was adopted.

Bed Sediment Porosity

The bed sediment porosity describes the volume of water per volume of benthic space. Bed sediment porosity is calculated from bed sediment concentration and sediment density as follows (Addendum):

$$\theta_{bs} = 1 - \frac{BS}{\rho_s} \qquad \text{(D.5-}\mathbf{(26)}\text{)}$$

where

θ_{bs} = bed sediment porosity (L/L)
BS = bed sediment concentration = 1 kg/L = 1,000,000 mg/L
ρ_s = sediment density = 2.65 kg/L (a standard value for mineral materials).

This results in a value of 0.6 that was used in the modeling simulations.

Gas-Phase Transfer Coefficient

The gas-phase transfer coefficient is used to estimate volatile losses from the waterbody. Volatile losses are calculated using a two-layer resistance model that incorporates a gas-phase transfer coefficient and a liquid-phase transfer coefficient. Both transfer coefficients are controlled by flow-induced turbulence in flowing systems. The liquid-phase transfer coefficient is calculated based on chemical-specific properties as specified in the Addendum. The Addendum recommends a single value for the gas-phase transfer coefficient for flowing systems of 36,500 m/yr.

There is some uncertainty around setting this parameter to a single value that is not chemical specific. It is reasonable to assume that chemical properties affecting volatility would have some effect on this value, although it is not known how large such an effect would be. The Addendum does give an equation (using chemical-specific properties) for calculating this parameter for stagnant systems, such as lakes or ponds. However, the transfer coefficients for stagnant systems are dominated by wind-induced turbulence rather than flow-induced turbulence; therefore, this equation is not applicable to flowing systems such as are modeled here and is not used.

Fraction Organic Carbon in Bottom Sediment

The fraction organic carbon in bottom sediment is derived from the fraction organic carbon in watershed soils. This value is “site-specific” and the same value is used for the waterbody as is used for the nonwaterbody soils.

Waterbody Temperature

An average surface waterbody temperature of 298 K (25°C) was considered a "common assumption for water temperature" in the Addendum. Although this value is somewhat high, the results are insensitive to this parameter, and reasonable lower values should have no effect on the results. This temperature was used to estimate gaseous diffusion loads into the surface waterbody.

Fish Concentrations

The chemical constituents in the water column exist in both the freely dissolved phase and sorbed to suspended solids or dissolved organic carbon. For all constituents, the dissolved fraction is the most bioavailable form and is used to predict the fish concentrations whenever biological uptake data (i.e., BCFs, BAFs, or BSAFs) are available for the dissolved phase of the constituent. In some cases (e.g., for metals), the biological uptake data are taken from studies that considered only the total concentration of the chemical, that is, the dissolved phase *and* the sorbed phase. For these constituents, the fish concentrations are predicted using a total chemical concentration rather than the freely dissolved concentration. The estimated surface water concentrations (see Appendix E for equations) are used to predict the concentration of contaminants in fish by applying constituent-specific biological uptake factors.

One of three types of biological uptake factors is used in the calculation of fish tissue concentrations: bioconcentration factors (BCFs), bioaccumulation factors (BAFs), and biota-sediment accumulation factors (BSAFs). Bioconcentration is defined as the net uptake of a chemical from an organism's surrounding medium through direct contact (e.g., uptake by a fish through the gill exchange) but excluding ingestion of contaminated plants or prey. Bioaccumulation is defined as the net uptake of a chemical from the environment from all pathways (including direct contact and ingestion of contaminated plants or prey). It is important to recognize that the distinction between BCF and BAF has both practical and technical implications. The route of exposure assumed for BCFs is direct contact, and BCF values are typically generated from controlled laboratory studies where fish are exposed to the chemical only through water. For organic chemicals with log K_{ow} values below ~4.0 (referred to here as hydrophilic organics), the BCF provides a reasonable estimate of the concentration expected to be found in fish under field conditions. However, for more hydrophobic organic chemicals (log K_{ow} >> 4.0), uptake via the food chain will be an increasingly important source of exposure, and using a BCF will tend to underestimate the concentration in fish tissue. Therefore, for hydrophobic organic chemicals, and other chemicals shown to bioaccumulate (e.g., mercury), a BAF is the preferred factor to use for estimating fish tissue concentrations. BAFs are typically generated from field studies or estimated from models.

In addition to the distinction between BCFs and BAFs, it is important to recognize the difference between dissolved water concentrations versus total water concentrations. As noted above, the surface water concentrations for hydrophilic organics are typically regarded as freely dissolved, although a small fraction will be sorbed to dissolved organic carbon (DOC) and larger organic particles. For hydrophobic organic chemicals, a significant fraction of the total water concentration is sorbed to DOC and particles. Unless filtration methods were used to analyze the freely dissolved concentration in the water, the reported biological uptake factors may reflect total rather than the dissolved water concentration. Thus, it is important to select the appropriate water concentration (freely dissolved or total) depending on the biological uptake factor that is available. For hydrophilic organics, we generally used the freely dissolved water concentration because the study-generated BCFs and BAFs (as well as the empirical algorithms) reflect the dissolved concentration in surface water. In contrast, we typically use the total concentrations for metals and hydrophobic organics because the BCFs and BAFs often are based on total rather than freely dissolved water concentrations.

The following equations are used to estimate fish tissue concentrations for constituents of concern other than dioxins:

$$C_{fish} = C_w \ x \ BAF \qquad \text{(D.5-}\mathbf{(27)}\text{)} \quad \text{or}$$

$$C_{fish} = C_w \ x \ BCF \qquad \text{(D.5-}\mathbf{(28)}\text{)}$$

where

C_{fish} = fish concentration (mg/kg)
C_w = water concentration - dissolved or total (mg/L)
BAF = bioaccumulation factor - dissolved or total (L/kg)
BCF = bioconcentration factor - dissolved or total (L/kg).

BSAFs were used to predict the dioxin concentration in fish tissue. The BSAF is a similar measure of uptake to BCFs or BAFs in that it is the ratio of the fish tissue concentration to the medium concentration; however, it is calculated from the sediment concentration rather than the surface water concentration. The calculation assumes that equilibrium has been reach among the sediment, pore water, and water column.

In freshwater systems, dioxins bioaccumulate in fish even though concentrations in the water column may be below the detection limit. Indeed, the analytical measures for highly hydrophobic constituents are much more reliable for sediments than for surface waters, and sediment-based uptake factors are generally preferred. Therefore, the BSAF was used to predict fish tissue concentrations for dioxin congeners using the following equation:

$$C_{fish} = \frac{C_{sed} \; x \; BSAF \; x \; f_{lipid}}{f_{oc}} \qquad \text{(D.5-}\mathbf{(29)}\text{)}$$

where

C_{fish} = fish tissue concentration (mg/kg)
C_{sed} = sediment concentration (mg/kg)
BSAF = biota-sediment bioaccumulation (kg/kg)
f_{lipid} = lipid content of fish (unitless)
f_{oc} = fraction organic carbon in bottom sediment (unitless).

References

U.S. EPA (Environmental Protection Agency). 1993. *Addendum to Methodology for Assessing Health Risks Associated with Indirect Exposure to Comubstor Emissions*. EPA/600/AP-93003. Office of Health and Environmental Assessment, Washington, DC.

Van der Leeden, F., F.L. Troise, and D.K. Todd. 1990. *The Water Encyclopedia*. 2nd ed. Chelsea, MI: Lewis Publishers.

Appendix F.1

Monte Carlo Fate and Transport Modeling

Appendix F.1

Monte Carlo Fate and Transport Modeling

Introduction

The probabilistic analysis addresses only constituents and pathways that had an excess lifetime cancer risk of 10^{-6} or greater or a hazard quotient of 1 or greater. Spreadsheet models, computer programs, and Crystal Ball™ version 4.0d were used to run the Monte Carlo analysis. This document describes how the common input parameters, their distributions, and the transfer of input-output data were handled.

Eight waste quantities were used in the probabilistic assessment for wastewaters managed in tanks (Table 2-4) and nine wastewater sludge quantities were used for the landfills (Table 2-6). Only one waste quantity was used for the land treatment unit (LTU) because only one facility uses an LTU. Waste quantity and facility location were independent variables in the risk assessment.

For the deterministic analysis, an average waste concentration and a maximum concentration were used to represent the central tendency and high-end concentrations, respectively. For the Monte Carlo analysis, waste concentrations were determined by selecting one of the available samples at random. Each sample (see Tables 2-1, 2-2, and 2-3) had an equal probability of being selected. Constituents of concern (COCs) for the LTU included dioxins (nongroundwater pathways), arsenic (nongroundwater and groundwater pathways), and chromium (direct inhalation). Arsenic (groundwater ingestion) was the only COC for the landfill. Dioxins were the only COCs for wastewaters.

The key distinction between deterministic and Monte Carlo analyses is that for the deterministic analysis, each input parameter is set at either a central tendency or high-end value with a maximum of two parameters set at the high-end value at any time. For the Monte Carlo analysis on the other hand, probability density functions (PDFs) are developed that describe the full range of values for the various input parameters. The result from the Monte Carlo analysis is a probability distribution of exposures and risk, rather than a single value. PDFs for input parameters used to model constituents released from EDC/VCM wastewater sludges managed in a landfill are shown in Figure F.1-1 (unsaturated zone), Figure F.1-2 (saturated zone), and Figure F.1-3 (source specific parameters). Unsaturated zone, saturated zone, and source specific input PDFs used to model the LTU are shown in Figures F.1-4, Figure F.1-5, and F.1-6, respectively. PDFs are not shown for waste concentration (Tables 2-1 to 2-3), waste quantities (Tables 2-4 and 2-6), location (Tables 2-5 and 2-6), or distance to receptor (50, 75, 100, 200, 300, 500, and 1000 m) because values were selected with equal probability from a limited number of available or selected values rather than a continuous distribution. The various input parameters used in the Monte Carlo analysis are described in Appendix K.

For the non-groundwater pathway analyses, fate and transport modeling and exposure modeling were linked together and are discussed in more detail in Appendix F.2. For groundwater pathways, the fate and transport models are separate from the exposure modeling. A PDF for groundwater concentrations at the groundwater exposure point (receptor well) is generated from the EPACMTP model. PDFs for groundwater concentrations generated for the landfill and LTU are shown in Figures F.1a and F.1-7b, respectively. The groundwater concentration PDF is then combined with exposure parameters (discussed in Appendix F.2) to generate a distribution of risk for each receptor. Therefore, in order to conduct a probabilistic risk analysis for groundwater exposure pathways, it was necessary to link output from the partitioning model with the groundwater model. The following sections describe the Monte Carlo methodologies used for groundwater fate and transport modeling.

Groundwater Pathway Monte Carlo Modeling

Detailed background for the Monte Carlo option in EPACMTP is provided in U.S. EPA 1997a and b. In order to run EPACMTP in Monte Carlo mode, a probability distribution must be provided for each input parameter (except constant or derived parameters). The Monte Carlo methodology is then performed as follows. The model input data are read first, followed by generation of a sequence of random numbers. This sequence of random numbers is then used to generate the pseudo-random values of the uncertain input variable(s) (drawn from the known distribution and within the range of any imposed bounds). The derived parameter values are then calculated. Following this, the contaminant fate and transport simulation is performed. The result is given in terms of the predicted contaminant concentrations in a downgradient receptor well. The generation of random parameter values and the fate and transport simulation is then repeated as many times as desired to generate a set of resulting receptor well concentrations. This model output is then statistically analyzed to yield the cumulative probability distribution of groundwater concentrations. A key aspect of the Monte Carlo analysis is the preservation of correlations between various model parameters, e.g. between waste site location and infiltration and recharge rate, and between location and hydrogeologic parameters. In order to maintain these dependencies in the analysis, a regional site-based modeling database has been developed (U.S. EPA, 1997a,b), which associates any waste unit location in the continental United States with one of 13 hydrogeologic regions and one of 97 climatic regions and provides modeling parameter values for each of these hydrogeological and climate regions. This database utilizes the U.S. Geological Survey (USGS) inventory of the groundwater resources of each state (Heath, 1984), and the Hydrogeologic Database for Modeling (HGDB) (API, 1989; U.S. EPA, 1997a and

1997b), developed from a survey of hydrogeologic parameters for actual hazardous waste sites in the United States.

Groundwater Pathway Monte Carlo Modeling for Land Treatment Units

Since there is only one Chlorinated Aliphatics facility that land applies their EDC/VCM sludge, only one waste unit location and waste volume were used for the Monte Carlo analysis.

In the present analysis, the calculation of leachate concentrations for the land treatment unit, and calculation of landfill volatilization losses, respectively (see Appendix D.1), were performed outside of the EPACMTP model. In order to ensure conservation of mass between the source partitioning model and EPACMTP groundwater pathway model, the models must be fully synchronized so that the same values for common input parameters are used in each individual Monte Carlo realization. In the Monte Carlo modeling, the output from the source partitioning model which is used as input for the groundwater modeling thus includes values for parameters such as waste unit area, and infiltration rate.

To implement the required Monte Carlo linkage between the land treatment unit partitioning model and EPACMTP, the data output routines of RTI's partitioning model were modified to generate output files that provide not only the leachate concentrations for each realization, but also the corresponding input values of common modeling parameters. For the Chlorinated Aliphatics analysis, these parameters are

- Waste Management Unit Area
- Infiltration Rate
- Site Location (in terms of hydrogeologic and climatic region)
- Maximum 9-year Average Leachate Concentration.

For the land treatment unit scenario, the source partitioning model was used to calculate the leachate concentration of arsenic (the only constituent modeled in the Monte Carlo analysis) infiltrating to the subsurface. This source model was run in Monte Carlo mode for 1,000 realizations to generate a probability distribution of leachate concentrations, creating a land treatment unit source data file of 1,000 records. This data file was then used as input to the EPACMTP groundwater fate and transport model. For this purpose, the EPACMTP data input routines were modified to read in the values of Monte Carlo parameters that are common to both models from this data file rather than generate these parameter values internally. For each realization of the subsequent groundwater pathway Monte Carlo analysis, EPACMTP picked a random record (with replacement) from the appropriate source file and read in the values of the common input parameters in that record. The hydrogeologic region and climate region indices that are specific to the LTU waste unit site were employed to choose random values for other saturated and unsaturated zone parameters appropriate to the waste unit location associated with the chosen record. For each of the 2,000 EPACMTP Monte Carlo realizations, the receptor well concentration of interest, e.g., maximum 9-year average concentration, was then calculated and saved to an output file. At the conclusion of the modeling, a PDF for the maximum 9-year average receptor well concentration was constructed.

For the landfill Monte Carlo modeling, a database of Chlorinated Aliphatics facility locations was created. Each facility location was assigned a hydrogeologic region number, a climate region number, a groundwater temperature, and an infiltration rate, from the OSW nationwide groundwater modeling database (U.S. EPA, 1997a,b). The landfill source partitioning was applied to generate a distribution of the mass volatilized during the active life of the landfill. The output of this model was a source data file of 1,000 records; each record contained the following data: landfill area, landfill depth, infiltration rate, waste concentration, initial leachate (TCLP) concentration, groundwater temperature, climate region number, and hydrogeologic region number. The leachate concentrations were chosen from TCLP sampling data.

To implement the required Monte Carlo linkage between the landfill volatilization model and EPACMTP, the data output routines of the volatilization model were modified in a manner similar to the land treatment unit partitioning model. That is, the output files provide not only the leachate concentrations for each realization, but also the corresponding input values of common modeling parameters.

For the first realization of the volatilization model, one value was chosen for each model input and the source partitioning equations were then used to generate the amount of mass volatilized. This mass volatilized and the input values for all common modeling parameters were saved as one record of an ASCII data file. This process was then repeated 1,000 times, producing an output data file containing 1,000 records. The waste concentration was then corrected for the volatilization mass loss and added to the data file. This data file was then used as input to the EPACMTP groundwater fate and transport model.

For each realization of the subsequent groundwater pathway Monte Carlo analysis, EPACMTP picked a random record (with replacement) from the appropriate source file; the model then read in the values of the common input parameters in that record. The hydrogeologic region and climate region indices that are specific to each waste unit site were then employed to choose random values for other saturated and unsaturated zone parameters appropriate to the waste unit location associated with the chosen record. For each of the 2,000 EPACMTP Monte Carlo realizations, the receptor well concentration of interest, e.g., maximum 9-year average concentration, was then calculated and saved to an output file. At the conclusion of the modeling, a PDF for the maximum 9-year average receptor well concentration was constructed.

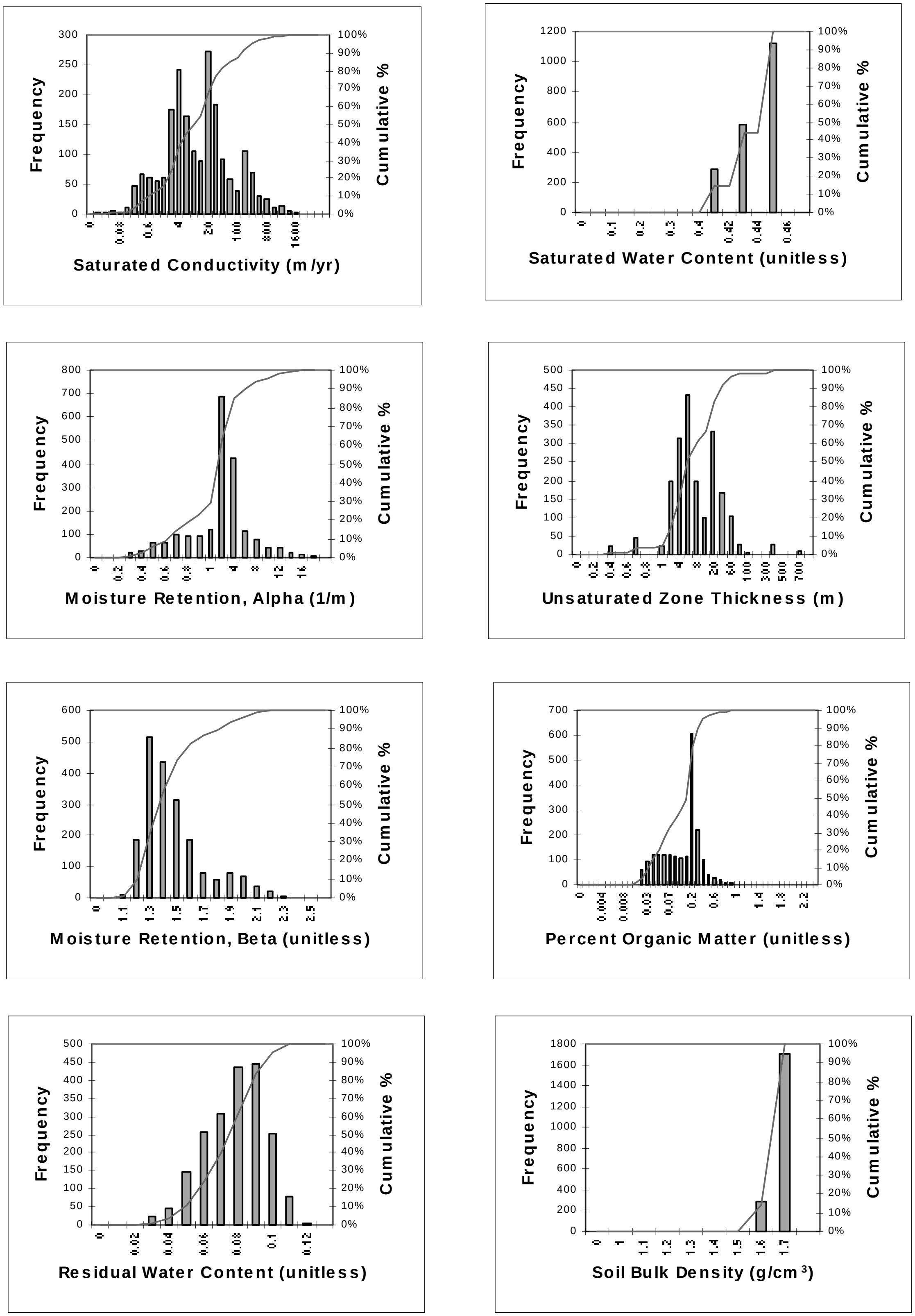


Figure F.1-1. Monte Carlo Input Parameters, Landfill - Unsaturated Zone

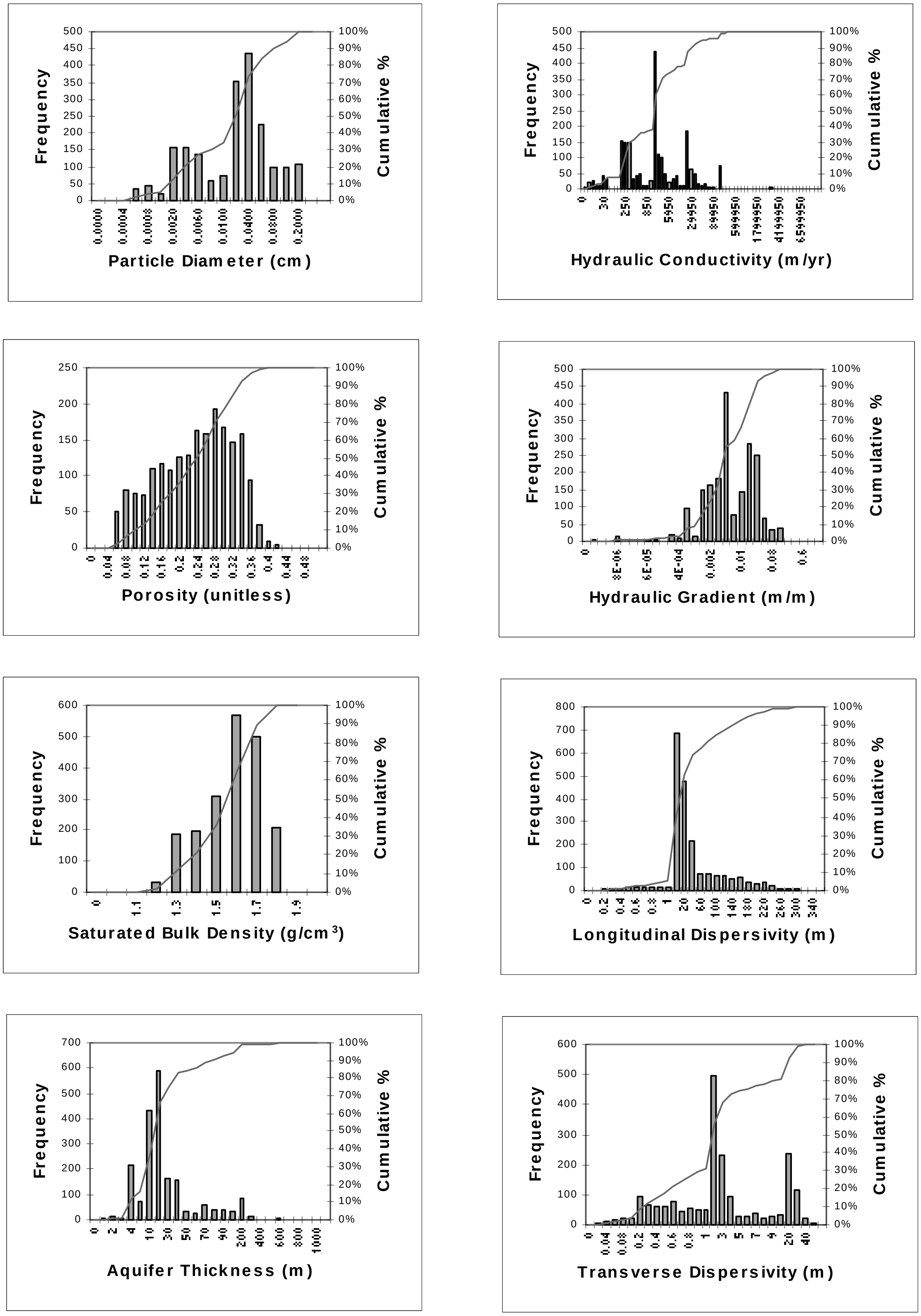


Figure F.1-2. Monte Carlo Input Parameters, Landfill -Saturated Zone

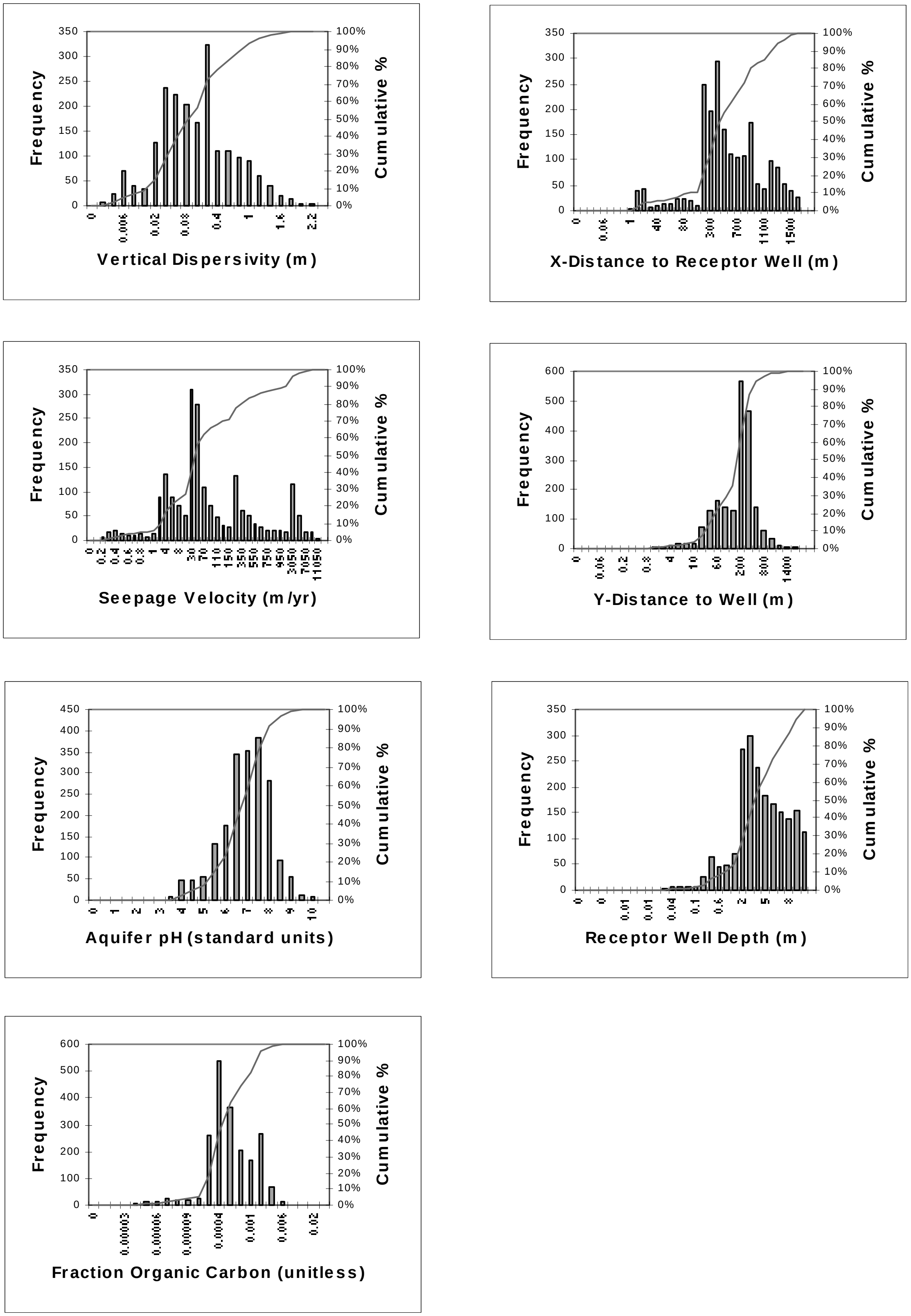
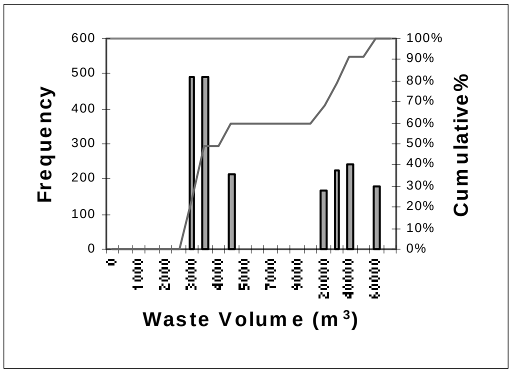
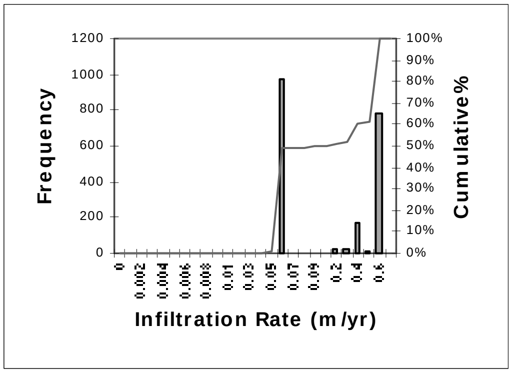
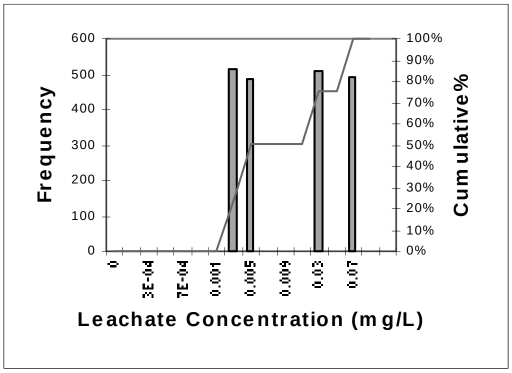
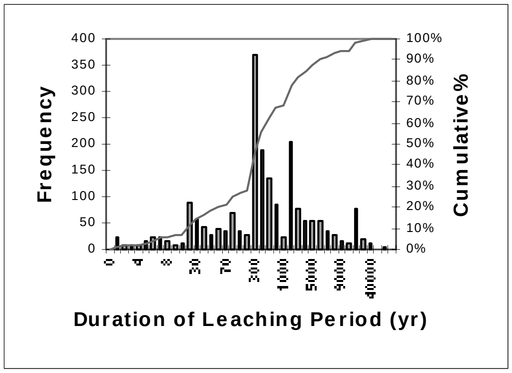


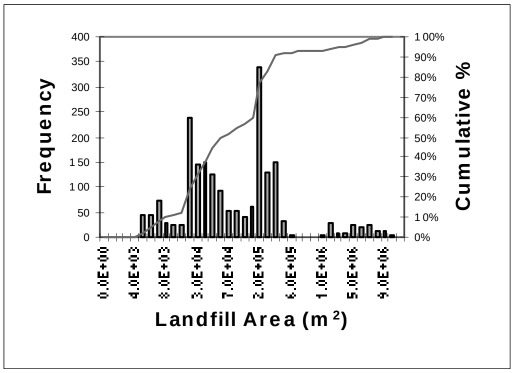
Figure F.1-2. Continued

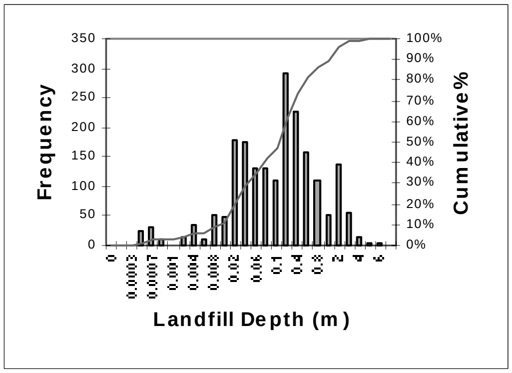












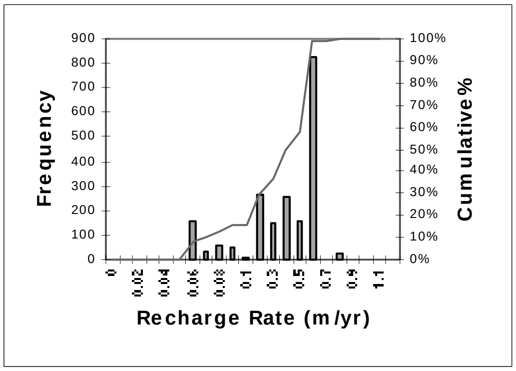


Figure F.1-3. Monte Carlo Source Specific Input Parameters - Landfill

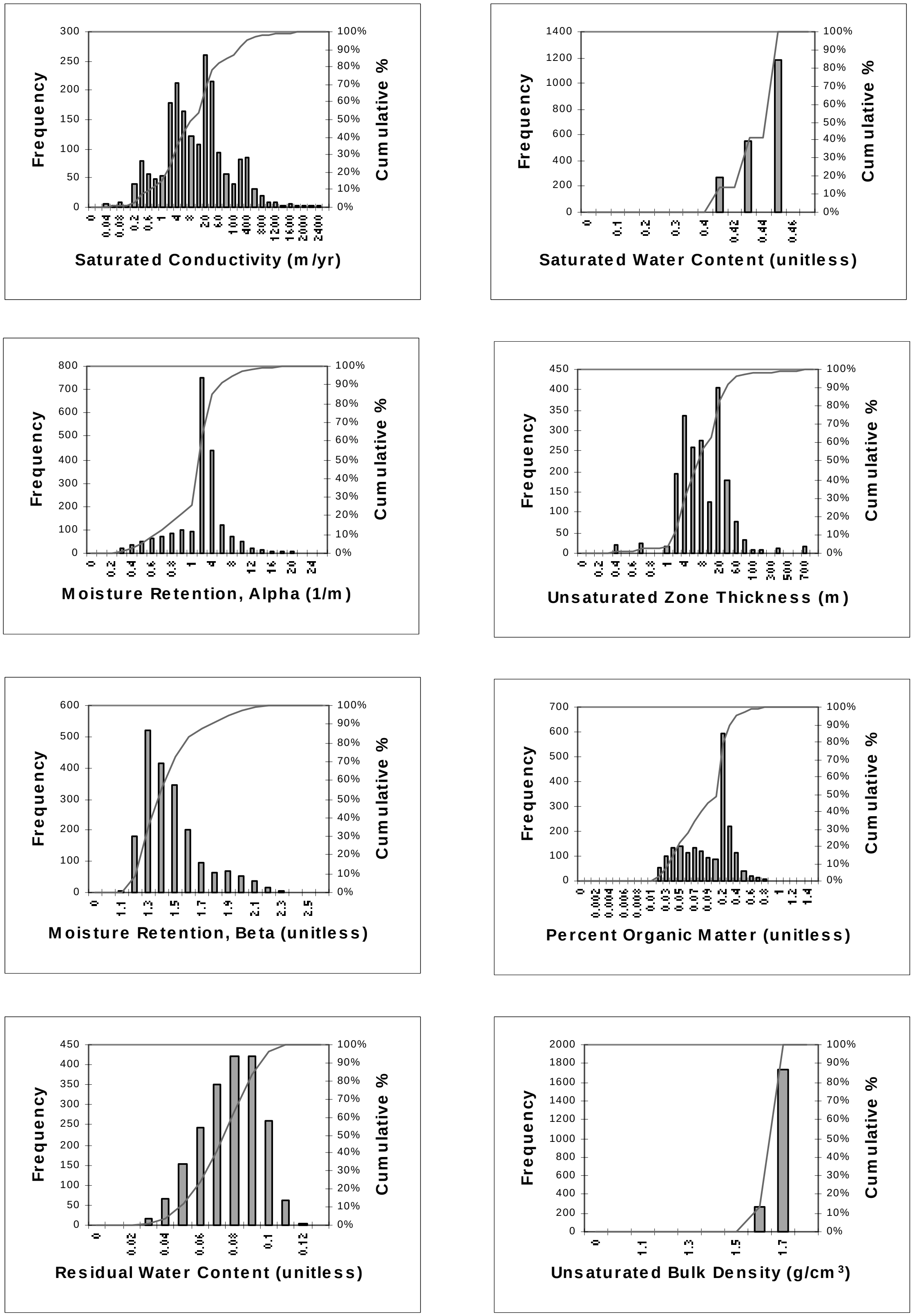


Figure F.1-4. Monte Carlo Input Parameters, Land Treatment Unit - Unsaturated Zone

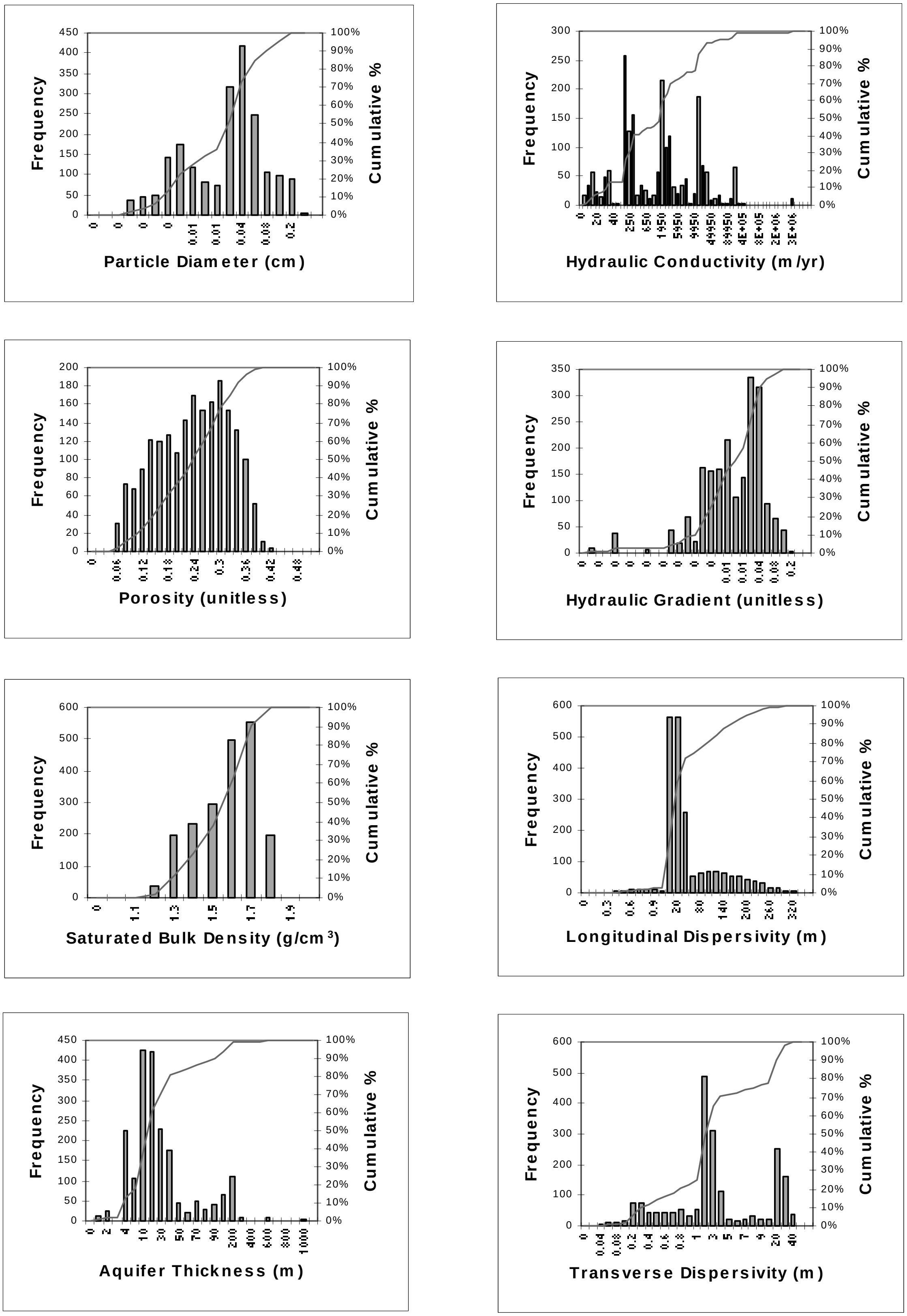


Figure F.1-5. Monte Carlo Input Parameters, Land Treatment Unit - Saturated Zone

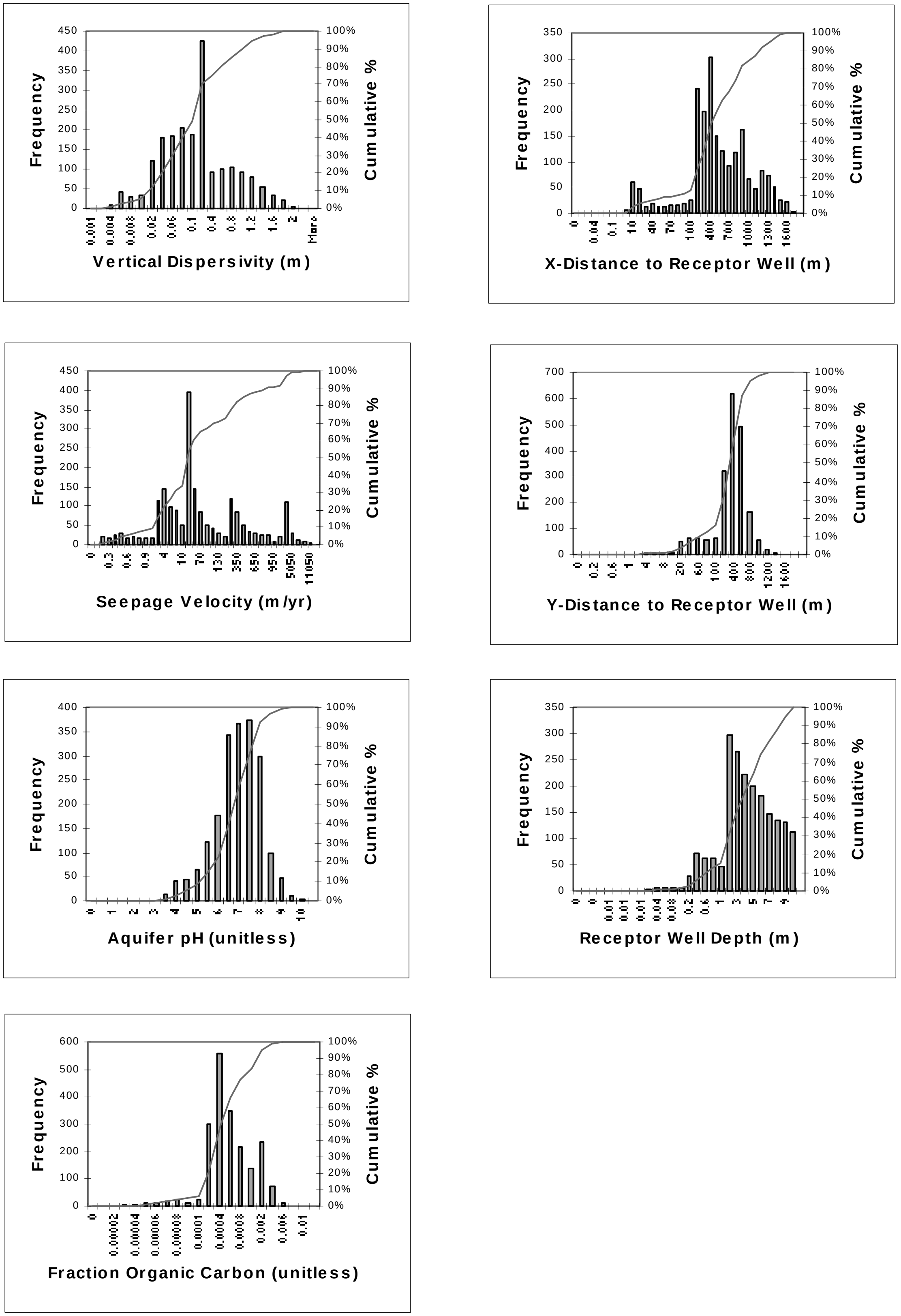


Figure F.1-5. (continued)

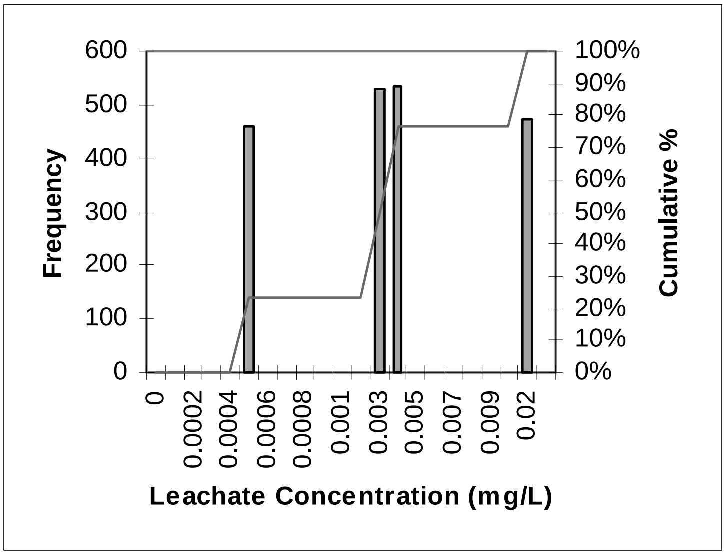


Figure F.1-6. Monte Carlo Source Specific Parameters - Land Treatment Unit

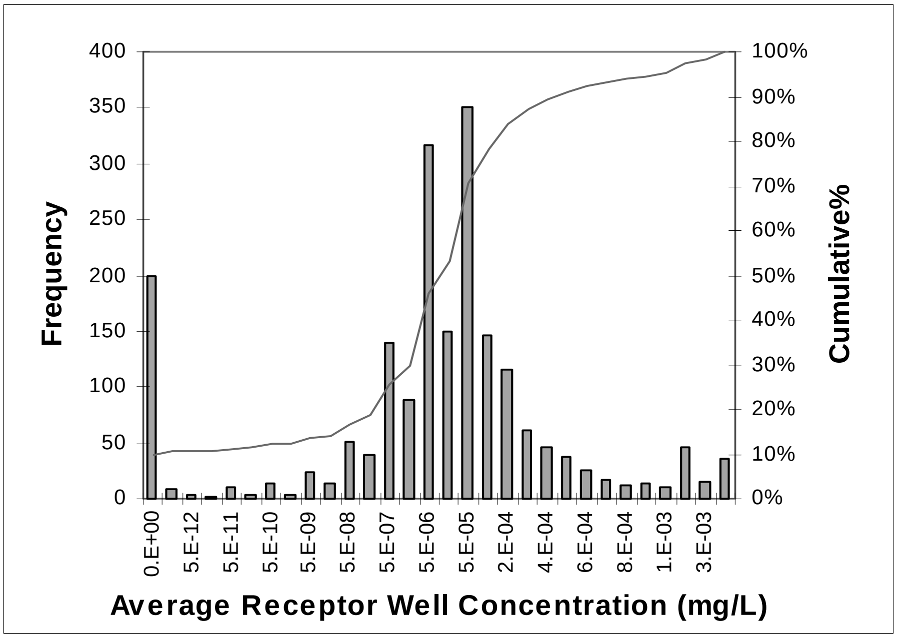


Figure F.1-7a. Chlorinated Aliphatics, Landfill Scenario , Monte Carlo Output Distribution for Average Receptor Well Concentrations for Arsenic

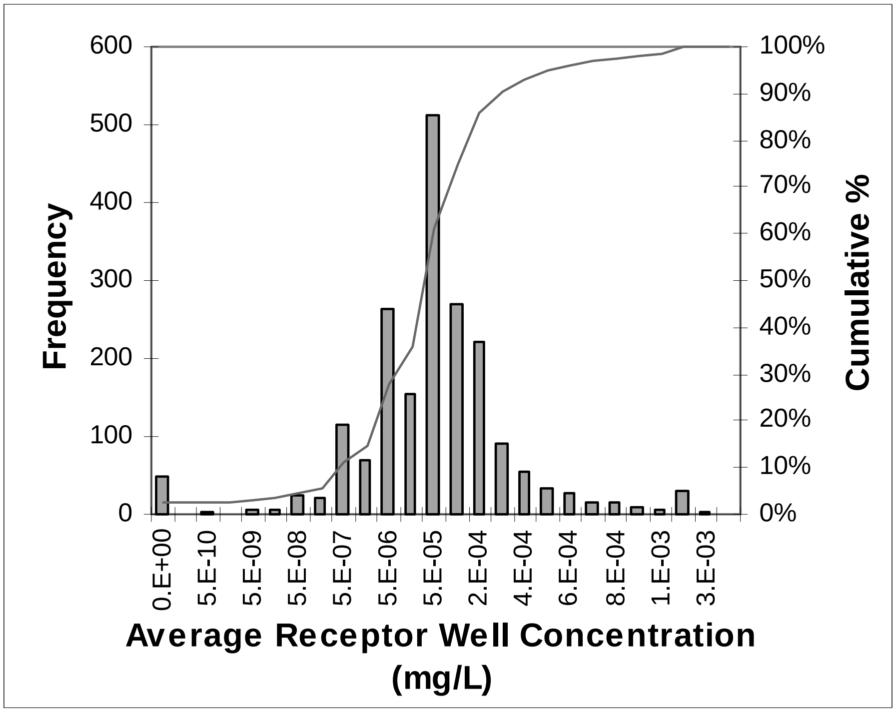


Figure F.1-7b. Chlorinated Aliphatics, Land Treatment Unit , Monte Carlo Output Distribution for Average Receptor Well Concentrations for Arsenic

References

API (American Petroleum Institute), 1989. *Hydrogeologic Database for Groundwater Modeling*. API Publication No. 4476, American Petroleum Institute.

Heath, R.C. 1984. *State Summaries of Groundwater Resources*. United States Geological Survey Water-Supply Paper 2275.

U.S. EPA (Environmental Protection Agency). 1997a. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) User's Guide*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997b. *Analysis of Subtitle D Waste Management Units Databases Used in the Groundwater Pathway Analysis for HWIR*. Office of Solid Waste, Washington, DC.

8. ERRATA

The following sections present errors that we noted in the 1999 Risk Assessment Technical Background Document. None of these errors changes the outcome of the analysis, but the corrections are presented here for the sake of completeness. In cases where errors were noted in tables, we provide revised tables, and indicate the modified information by double-underline or by a double-lined cell border.

Section 2

1. p. 2-10. Table 2-2. The analytical result shown for the TCLP arsenic concentration for sample OC-02 is incorrect. The value should be shown as <0.0025, half of the detection limit (0.005), rather than as <0.005. The central tendency arsenic TCLP concentration should be calculated as 0.0186 mg/L, rather than 0.0193 mg/L. The correct concentrations were used in EPA's risk analyses, the values were only reported incorrectly on the table.

2. p. 2-11. Table 2-2. Our analyses always consider the concentrations of each of the specific congeners since the congeners have different fate and transport properties. Consequently, the results for "Total TCDF," "Total HxCDF," and "Total HpCDF" reported by the laboratory are superfluous and should be disregarded.

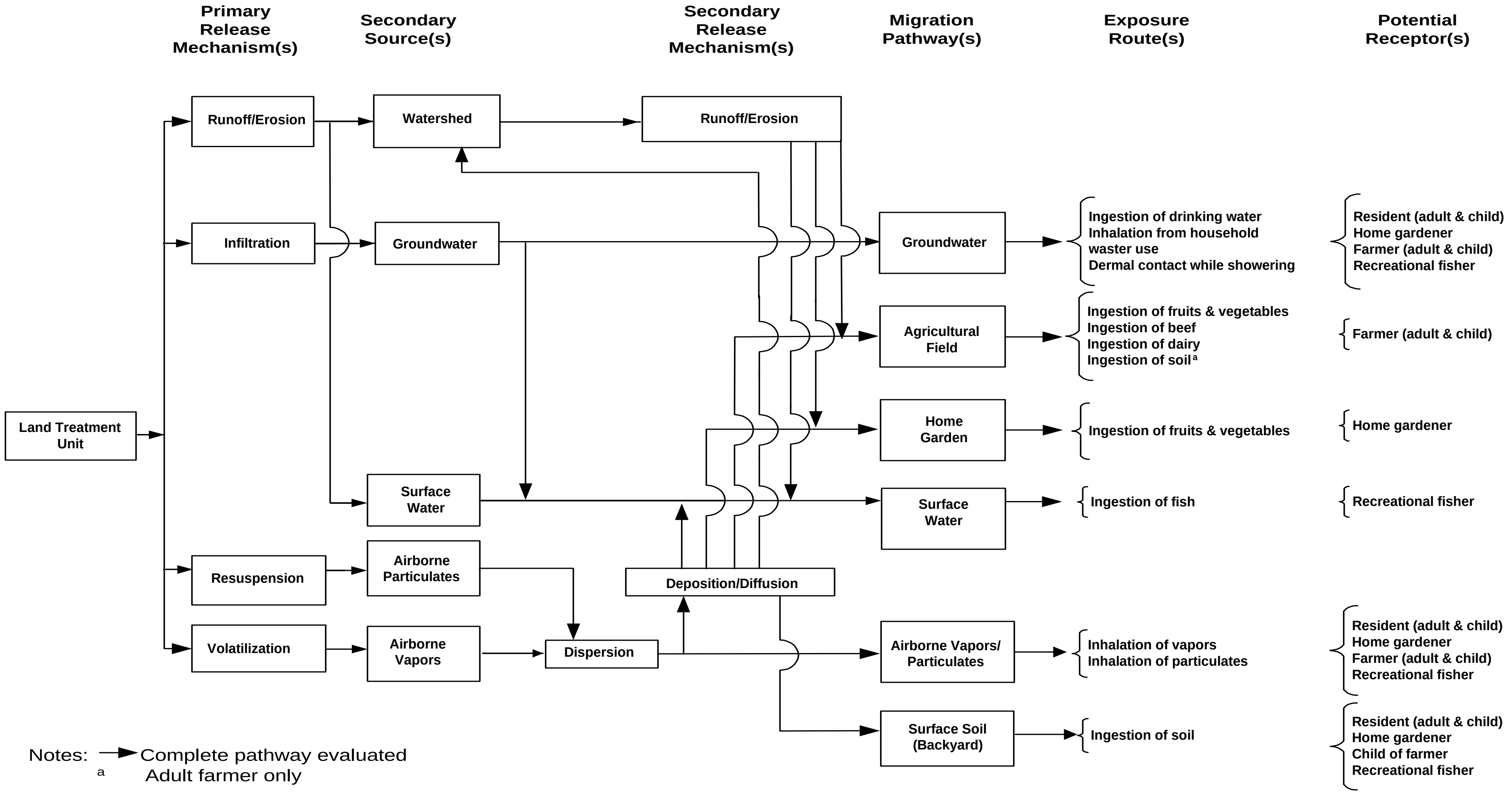
3. p. 2-26, 3rd paragraph. Although the specific infiltration rates that we used in our groundwater analyses are presented in Appendix K, this paragraph should clarify that in conducting the groundwater fate and transport modeling for the deterministic EDC/VCM landfill analysis, we used infiltration rate data from the Brownsville meteorological station to represent our central tendency site location, Houston. Brownsville was the meteorological station closest to Houston for which we had data. We also used the Brownsville infiltration rate data in lieu of the Houston meteorological station in our probabilistic groundwater pathway risk analyses. The Houston meteorological station was used as the central tendency location for all of our non-groundwater modeling. Houston subsurface parameters were used as central tendency in the groundwater modeling.

4. p. 2-32. Fisher Scenario. This section says that the distance to the stream was fixed at 102m from the source. However, stream distance actually was varied in the deterministic analyses (that is, stream distance was included as a sensitive parameter in the sensitivity analyses). The stream distances evaluated were 102m and 300m for the chlorinated aliphatics wastewater tank, EDC/VCM landfill, and methyl chloride landfill scenarios, and 102m and 403m for the EDC/VCM land treatment unit scenario. The impact of the difference in the analysis as described and as implemented is provided in more detail in the discussion of Appendix D.5, below.

5. p. 2-36. Figure 2-5. Some information was left off of the figure as it appeared in the 1999 document. The revised figure is presented on the following pages.

6. p. 2-39, 3rd bullet. Manganese should be deleted from the list of constituents we did not evaluate because they are essential nutrients. Manganese was included in our analyses.

7. pp. 2-40 through 2-43. Table 2-8. Several errors were noted in Table 2-8. A revised version of Table 2-8 is provided on the following pages.



Notes: → Complete pathway evaluated
[a] Adult farmer only

Figure 2-5 (Revised). Exposure Pathways - Wastewater Treatment Sludge Managed in an On-Site Land Treatment Unit

Table 2-8 (REVISED). COPCs for Chlorinated Aliphatics Wastewaters, EDC/VCM Sludges, and Methyl Chloride Sludges

Constituent	CAS No.	Chlorinated Aliphatics Wastewaters	EDC/VCM Sludges				Methyl Chloride Sludges	
			Groundwater Pathways		Nongroundwater Pathways			
		Non-Groundwater Pathways	LTU	Landfill	LTU	Landfill	Groundwater Pathways	Non-Groundwater Pathways
Volatile Organics								
1,2-Dichloroethane	107062	**X**	g	**X**	**X**	**X**		
2-Chloro-1,3-butadiene	126998	**X**						
Acetone	67641	**X**	g	g	**X**	**X**	g	**X**
Allyl chloride	107051	**X**	g		**X**	**X**		
Bromoform	75252	j*	g					
Carbon disulfide	75150	**X**	g	g	**X**	**X**	g	
Chlorobenzene	108907	**X**						
Chloroethane	75003	**X**						
Chloroform	67663	**X**	g	**X**	**X**	**X**		
cis-1,2-Dichloroethylene	156592	**X**						
cis-1,3-Dichloropropene	10061015			**X**				
Chlorodibromomethane	124481	j*						
Ethylbenzene	100414	**X**						
2-Hexanone	591786		j		**j**	**j**		
Methyl ethyl ketone	78933	**X**	g	g	**X**	**X**		
Methylene chloride	75092	j*	g	**X**	**X**	**X**	g	**X**
Styrene	100425	**X**						
Tetrachloroethylene	127184	**X**	g,j		j*	j*		
trans-1,2-Dichloroethylene	156605	j*						
Trichloroethylene	79016	**X**	g,j		j*	j*		
Vinyl Chloride	75014		g,j		j*	j*		
Vinyl Acetate	108054		g		**X**	**X**		

(continued)

X = Constituent of Potential Concern.
g = Constituent screened out of the groundwater pathway analysis.
v = Constituent screened out of the nongroundwater pathway analysis (only the vapor air pathway was evaluated).
e = Essential nutrient.
j = All values were non-detect, with the exception of one “J”-qualified (estimated) value; “*” indicates that although screened out, the constituent was retained in the analysis inadvertently.

Table 2-8 (REVISED). (continued)

Constituent	CAS No.	Chlorinated Aliphatics Wastewaters	EDC/VCM Sludges				Methyl Chloride Sludges	
		Non-Groundwater Pathways	Groundwater Pathways		Nongroundwater Pathways		Groundwater Pathways	Non-Groundwater Pathways
			LTU	Landfill	LTU	Landfill		
Semivolatile Organics								
4-Aminobiphenyl	92671	j						
Benzoic acid	65850	**X**	g,j	g	j*	j*	g	
Benzyl alcohol	100516	**X**					g	
Bis(2-chloroethyl)ether	111444	**X**	**X**	**X**	**X**	**X**		
Bis(2-chloroisopropyl)ether	39638329	**X**						
1,4-Dioxane	123911		**X**	**X**				
2-(2-Chloroethoxy)ethanol	628897		**X**	**X**				
Diethyl phthalate	84662	**X**						
Dimethyl phthalate	131113	j						
Bis(2-ethylhexyl)phthalate	117817	j*	g		**X**	**X**		
Hexachlorobenzene	118741	j*	g,j		j*	j*		
4-Methyl-2-pentanone	108101			g				
2-Methylphenol	95487	**X**						
4-Methylphenol	106445	**X**		g				
Di-n-octyl phthalate	117840	j*						
Pentachlorophenol	87865	**X**						
Phenol	108952	**X**						
2,4,5-Trichlorophenol	95954	**X**						
2,4,6-Trichlorophenol	88062	**X**						
Metals								
Aluminum	7429905	v	**X**		**X**	v		v
Arsenic	7440382	v	**X**	**X**	**X**	v	**X**	v

(continued)

X = Constituent of Potential Concern.
g = Constituent screened out of the groundwater pathway analysis.
v = Constituent screened out of the nongroundwater pathway analysis (only the vapor air pathway was evaluated).
e = Essential nutrient.
j = All values were non-detect, with the exception of one “J”-qualified (estimated) value; “*” indicates that although screened out, the constituent was retained in the analysis inadvertently.

Table 2-8 (REVISED). (continued)

Constituent	**CAS No.**	**Chlorinated Aliphatics Wastewaters**	**EDC/VCM Sludges**				**Methyl Chloride Sludges**	
		Non-Groundwater Pathways	**Groundwater Pathways**		**Nongroundwater Pathways**		**Groundwater Pathways**	**Non-Groundwater Pathways**
			LTU	**Landfill**	**LTU**	**Landfill**		
Barium	7440393	v	g		**X**	v		
Beryllium	7440417	v						
Cadmium	7440439	v	g		**X**	v		
Calcium	7440702	v	e	e	e	v	e	v
Chromium	7440473	v	g		**X**	v		v
Cobalt	7440484	v	g	g	**X**	v		
Copper	7440508	v	**X**	**X**	**X**	v	**X**	v
Iron	7439896	v	e		e	v		v
Lead	7439921	v	**X**		**X**	v		v
Magnesium	7439954	v	e	e	e	v	e	v
Manganese	7439965	v	g	X	X	v	g	v
Mercury	7439976	**X**						
Molybdenum	7439987	v	g	**X**	**X**	v		
Nickel	7440020	v	g	**X**	**X**	v		v
Potassium	7440097	v	e	e	e	v		
Sodium	7440235	v	e		e	v		
Vanadium	7440622	v	g		**X**	v		
Zinc	7440666	v	g	g	**X**	v	g	v
Dioxins/Furans								
1,2,3,4,6,7,8-HpCDD	35822469	**X**	g		**X**	**X**		**X**
1,2,3,4,6,7,8-HpCDF	67562394	**X**	g	**X**	**X**	**X**		**X**
1,2,3,4,7,8,9-HpCDF	55673897	**X**	g	**X**	**X**	**X**		
1,2,3,4,7,8-HxCDD	39227286	**X**	g		**X**	**X**		

(continued)

X = Constituent of Potential Concern.
g = Constituent screened out of the groundwater pathway analysis.
v = Constituent screened out of the nongroundwater pathway analysis (only the vapor air pathway was evaluated).
e = Essential nutrient.
j = All values were non-detect, with the exception of one "J"-qualified (estimated) value; "*" indicates that although screened out, the constituent was retained in the analysis inadvertently.

Table 2-8 (REVISED). (continued)

Constituent	CAS No.	Chlorinated Aliphatics Wastewaters	EDC/VCM Sludges				Methyl Chloride Sludges	
		Non-Groundwater Pathways	Groundwater Pathways		Nongroundwater Pathways		Groundwater Pathways	Non-Groundwater Pathways
			LTU	Landfill	LTU	Landfill		
1,2,3,6,7,8-HxCDD	57653857	**X**	g		**X**	**X**		
1,2,3,7,8,9-HxCDD	19408743	**X**	g		**X**	**X**		
1,2,3,4,7,8-HxCDF	70648269	**X**	g		**X**	**X**		
1,2,3,6,7,8-HxCDF	57117449	**X**	g		**X**	**X**		
1,2,3,7,8,9-HxCDF	72918219	**X**	g		**X**	**X**		
2,3,4,6,7,8-HxCDF	60851345	**X**	g		**X**	**X**		
2,3,4,7,8-PeCDF	57117314	**X**	g		**X**	**X**		
1,2,3,7,8-PeCDF	57117416		g		**X**	**X**		
2,3,7,8-TCDD	41903575	**X**	g		**X**	**X**		
2,3,7,8-TCDF	51207319	**X**	g		**X**	**X**		
OCDD	3268879	**X**	g	**X**	**X**	**X**		**X**
OCDF	39001020	**X**	g	**X**	**X**	**X**		**X**

X = Constituent of Potential Concern.

g = Constituent screened out of the groundwater pathway analysis.

v = Constituent screened out of the nongroundwater pathway analysis (only the vapor air pathway was evaluated).

e = Essential nutrient.

j = All values were non-detect, with the exception of one “J”-qualified (estimated) value; “*” indicates that although screened out, the constituent was retained in the analysis inadvertently.

Notes:

[1] OCDD was screened out of the groundwater pathway analysis for the landfill, but was retained in the analysis inadvertently.

Section 3

1. p. 3-1, 2nd paragraph. The last sentence should be changed to "The partitioning equations used to estimate volatilization from the source are based on equations presented in a series of articles by Jury et al. (Jury et al., 1983, 1984, and 1990)."

2. p. 3-1, 3rd paragraph, 1st sentence. “erosion and” should be deleted from the sentence because the source partition model does not predict mass losses from erosion.

3. p. 3-2, 3rd paragraph. The third sentence of the paragraph was truncated and only referenced three congeners for which we capped concentrations a their solubility concentrations, not four. Further, the sentence can be re-written for additional clarity, as follows: “Consequently, for both the deterministic and probabilistic analyses, we constrained the overall influent contaminant load to the wastewater treatment system based on the aqueous solubility of the four congeners with sample concentrations that exceeded their respective solubility limits: 1,2,3,4,6,7,8-HpCDF, OCDD, OCDF, and 1,2,3,4,7,8,9-HpCDF.

4. p. 3-6, Table 3-1b. 1) The heading for the third column should read “GL-02" where it reads “G(ng/L)”. 2) Some values in four of the table rows were incorrect; however, the values actually used in the analysis were correct, and the error does not have any actual impact on the computed results. 3) The “Note” at the bottom of the table should include 1,2,3,4,6,7,8-HpCDF in the list of the congeners that were capped at their solubility limits. A revised version of Table 3-1b is included on the following page.

5. p. 3-11, Table 3-2. “LF Area” refers to the “landfill area” being the high-end parameter of concern. X-well refers to the longitudinal distance to the well and Y-well refers to the lateral distance to the well from the plume center line. The column entitled “Waste Volume” indicates the waste volume in the landfill that was used in the fate and transport modeling for the particular model run that is defined by that row in the table. As indicated in the waste volume column, the central tendency waste volume is 15,202 m^3, and the high end value is 50,579 m^3.

6. p. 3-16, 3rd paragraph, 5th sentence. The MCL for mercury is 0.002 mg/L, not 0.005 mg/L.

Table 3-1b (Revised). Estimated High End Dioxin Congener Emissions from Wastewater Tanks

Congener	TEF	GL-02 Concentration (ng/L)	Annual Quantity (g/yr)[a]	Annual TEQ Quantity (g/yr)[a]	Annual Emissions (Mg/yr)	Annual Emissions (g/yr)	Annual TEQ Emissions (g/yr)
1,2,3,4,6,7,8-HpCDD	0.01	0.880	0.283	0.0028	4.03e-10	4.03e-04	4.03e-06
1,2,3,4,6,7,8-HpCDF	0.01	43.0	13.844	0.1384	6.92e-09	6.92e-03	6.92e-05
1,2,3,4,7,8,9-HpCDF	0.01	12.0	3.863	0.0386	6.92e-09	6.92e-03	6.92e-05
1,2,3,4,7,8-HxCDD	0.1	0.052	0.017	0.0017	7.25e-11	7.25e-05	7.25e-06
1,2,3,6,7,8-HxCDD	0.1	0.091	0.029	0.0029	2.65e-10	2.65e-04	2.65e-05
1,2,3,7,8,9-HxCDD	0.1	0.110	0.035	0.0035	3.20e-10	3.20e-04	3.20e-05
1,2,3,4,7,8-HxCDF	0.1	5.30	1.706	0.1706	1.80e-08	1.80e-02	1.80e-03
1,2,3,6,7,8-HxCDF	0.1	1.20	0.386	0.0386	1.80e-09	1.80e-03	1.80e-04
1,2,3,7,8,9-HxCDF	0.1	0	0	0	0	0	0
2,3,4,6,7,8-HxCDF	0.1	0.430	0.138	0.0138	1.05e-09	1.05e-03	1.05e-04
2,3,4,7,8-PeCDF	0.5	0.210	0.068	0.0338	5.96e-10	5.96e-04	2.98e-04
2,3,7,8-TCDD	1	0.017	0.005	0.0055	6.30e-11	6.30e-05	6.30e-05
2,3,7,8-TCDF	0.1	0.082	0.026	0.0026	5.77e-10	5.77e-04	5.77e-05
OCDD	0.001	6.90	2.221	0.0022	7.79e-14	7.79e-08	7.79e-11
OCDF	0.001	6,000	1931.676	1.9317	5.52e-11	5.52e-05	5.52e-08
		Total	**1954.300**	**2.387**		**0.037**	**0.003**
					% Total	**0.002%**	**0.11%**

[a] Based on central tendency annual waste quantity of 321,946 m^3 or metric tons.

Note: The TEQ emissions estimates presented in this table are based on the solubility limits for 1,2,3,4,6,7,8-HpCDF, 1,2,3,4,7,8,9-HpCDF, OCDD, and OCDF (see Appendix C), and the sample concentrations in the PL-01 and GL-02 samples for the other congeners.

Section 4

1. pp. 4-5 and 4-5, Tables 4-1 and 4-2. Some of the entries in the table were incorrect (rows of the table were shifted in the version of the table presented in the 1999 Risk Assessment TBD). Revised tables are provided on the following pages. The child exposure factors are correct as presented in Appendix K.

2. p. 4-10, 1st paragraph. The last two sentences of this paragraph are incorrect and should be deleted.

3. p. 4-13, Table 4-4. The high end concentrations for chlorinated aliphatics wastewaters were incorrect as presented in the 1999 Risk Assessment TBD (the concentrations presented correspond to a distance to receptor of 75m). The correct concentrations were presented in Table H.1-1a of Appendix H.1 in the 1999 document. In addition, two values were updated to be consistent with a revised version of the source document.

4. p. 4-17. Section 4.2.4. 2-(2-chloroethoxy)ethanol should have been included in the list of constituents for which no toxicity values are available.

5. p. 4-19, 2nd bullet. The word “and” after “sludge” should be deleted.

6. p. 4-19, 1st paragraph. The sentence: “The concentration of lead in chlorinated aliphatics wastewater exceeds this concentration, as well as the background concentrations reported by Hunter (1998)” should be deleted. In our analysis of wastewaters we were only concerned with vapor emissions. The evaluation of lead is not relevant to the analysis. The text should note, however, that the leachate concentration predicted for the land treatment unit based on sludge dry weight concentrations, 0.005 mg/L (USEPA, 1999, Appendix B, Table B-1), is well below the action level for lead in drinking water (0.015 mg/L).

Table 4-1 (Revised). Time-weighted Average (TWA) Intake Rates for Central Tendency Child Receptor

Parameter	Units	Data source[a]	Age Group 1-2	Age Group 3-5	Age Group 6-11	Age Group 12-19	Calculated TWA	TWA Units	Comments
Soil ingestion rate	kg/d	Table 4-23	0.0001	0.0001	0.00005	0.00005	2.70E-06	kg/kg/d	
Drinking water ingestion rate	L/kg/d	Tables 3-7 & 3-30	0.0305			0.0163	2.38E-02	L/kg/d	ages 1 - 10 and 11 - 19
Fruit ingestion rate	g DW/kg/d	Table 9-7 & 13-61	0.432	0.271	0.165	0.0907	1.79E-04	kg DW/kg/d	WW concentrations in EFH converted to DW
Vegetable ingestion rate	g DW/kg/d	Table 13-63	0.166	0.102	0.0566	0.0577	7.58E-05	kg DW/kg/d	WW concentrations in EFH converted to DW
Root ingestion rate	g DW/kg/d	Table 13-65	0.92	0.462	0.523	0.565	5.73E-04	kg DW/kg/d	WW concentrations in EFH converted to DW
Root ingestion rate	g WW/kg/d	calculated	0.131	0.066	0.074	0.080	8.16E-05	kg WW/kg/d	
Dairy ingestion rate	g DW/kg/d	Table 11-2	23.48	19.52	11.88	5.29	1.15E-02	kg DW/kg/d	WW concentrations in EFH converted to DW
Dairy ingestion rate	g WW/kg/d	calculated	5.635	4.685	2.851	1.270	2.77E-03	kg WW/kg/d	
Beef ingestion rate	g DW/kg/d	Tables 11-3 & 13-36	1.339	1.162	2.11	1.51	1.63E-03	kg WW/kg/d	WW concentrations in EFH converted to DW
Beef ingestion rate	g WW/kg/d	calculated	0.380	0.330	0.599	0.429	4.62E-04	kg WW/kg/d	
Inhalation rate	m^3/d	Myers et al. 1998, Table 5-2		5.4	8.6	12	3.15E-01	$m^3/kg/d$	ages 0 - 3, 4 - 10, 11 - 18
Average body weight for soil ingestion	kg	Tables 7-6 & 7-7	12.15	17.13	29.48	56.61		kg	used to calculate TWA for soil ingestion
Average body weight for inhalation	kg	Tables 7-6 & 7-7		12.13	24.91	53.55		kg	used to calculate TWA for inhalation

DW = dry weight
WW = wet weight

[a] Tables are found in the *Exposure Factors Handbook (EFH)*

Table 4-2 (Revised). Time-weighted Average (TWA) Intake Rates for High End Child Receptor

Parameter	Units	Data source[a]	Age Group 1-2	Age Group 3-5	Age Group 6-11	Age Group 12-19	Calculated TWA	TWA Units	Comments
Soil ingestion rate	kg/d	Table 4-23	0.0004	0.0004	0.00005	0.00005	8.06E-06	kg/kg/d	
Fruit ingestion rate	g DW/kg/d	Tables 9-7 & 13-61	2.263	0.806	1.040	0.5081	9.08E-04	kg DW/kg/d	WW concentrations in EFH converted to DW
Vegetable ingestion rate	g DW/kg/d	Table 13-63	0.942	0.305	0.2834	0.2068	3.24E-04	kg DW/kg/d	WW concentrations in EFH converted to DW
Root ingestion rate	g DW/kg/d	Table 13-65	10.4	4.73	5.59	3.32	5.00E-03	kg DW/kg/d	WW concentrations in EFH converted to DW
Root ingestion rate	g WW/kg/d	calculated	1.481	0.674	0.796	0.473	7.13E-04	kg WW/kg/d	
Dairy ingestion rate	g DW/kg/d	Table 11-2	45.72	39.54	25.38	12.75	2.44E-02	kg DW/kg/d	WW concentrations in EFH converted to DW
Dairy ingestion rate	g WW/kg/d	calculated	10.973	9.490	6.091	3.060	5.87E-03	kg WW/kg/d	
Beef ingestion rate	g DW/kg/d	Tables 11-3 & 13-36	2.783	3.163	11.4	3.53	5.88E-03	kg DW/kg/d	WW concentrations in EFH converted to DW
Beef ingestion rate	g WW/kg/d	calculated	0.790	0.898	3.238	1.003	1.67E-03	kg WW/kg/d	
Inhalation rate	m^3/d	Myers et al., 1998, Table 5-2		12.4	12.6	18.3	5.45E-01	$m^3/kg/d$	ages 0 - 3, 4 - 10, 11 - 18
Average body weight for soil ingestion	kg	Tables 7-6 & 7-7	12.15	17.13	29.48	56.61		kg	used to calculate TWA for soil ingestion
Average body weight for inhalation	kg	Tables 7-6 & 7-7		12.13	24.91	53.55		kg	used to calculate TWA for inhalation

DW = dry weight
WW = wet weight
[a] Tables are found in the *Exposure Factors Handbook (EFH)*

Table 4-4 (Revised). Comparison of Background Concentrations and Intake Rates of Dioxin to Concentrations and Intake Rates Associated with Chlorinated Aliphatics Wastes

Comparison of Dioxin Concentrations and Dioxin Daily Intakes: Background vs. Chlorinated Aliphatics Wastes

	Average Background		**Chlorinated Aliphatics Wastes (Adult Farmer Scenario)**			
	Source: "Estimating Exposure to Dioxin-Like Compounds" (U.S. EPA 1994)	**Source: "Dioxin Factsheet: Background Environmental Levels and Exposure" (U.S. EPA 1999, updated)**	**Chlorinated Aliphatics Wastewater (Incremental Risk)**		**EDC/VCM Land Treatment Unit**	
Dioxin Source			**Central Tendency**	**High End**	**Central Tendency**	**High End**
TEQ in waste			0.313 ng/L	7.414 ng/L	78.9 ng/kg	907 ng/kg
TEQ in soil	8 ng/kg (n=95)	Rural: 1 to 6 ng/kg Urban: 7 to 20 ng/kg	0.0011 ng/kg	0.007 ng/kg	5.4 ng/kg	61 ng/kg
TEQ in beef	0.29 ng/kg [0.48 ng/kg]* (n=14)	0.17 ng/kg	0.011 ng/kg	0.12 ng/kg	0.13 ng/kg	1.4 ng/kg
TEQ in dairy	***Milk*** 0 ng/kg [0.07 ng/kg] (n=2) ***Other Dairy*** 0.35 ng/kg [0.36 ng/kg] (n=5) ***Egg*** 0.0004 ng/kg [0.13 ng/kg] (n=8)	***Milk*** 0.03 ng/kg ***Other Dairy*** 0.1 ng/kg ***Egg*** 0.03 ng/kg	0.0032 ng/kg	0.03 ng/kg	0.030 ng/kg	0.32 ng/kg
TEQ in air	0.0949 pg/m^3 (n=84)	Rural: 0.002 to 0.02 pg/m^3 Urban: 0.02 to 0.2 pg/m^3	0.00032 pg/m^3	0.002 pg/m^3	NA	NA
TEQ Dioxin Intake	mean: 0.062 ng/d [0.119 ng/d]	0.070 ng/d WHO TEQ due to dioxins,furans,PCBs	0.0012 ng/d*	0.012 ng/d*	0.012 ng/d*	0.13 ng/d*

* Based on central tendency intake rates (and assumptions regarding the fraction of a farmer's food that is home-grown).

n = number of samples
NA = not calculated
[ ] = For 1994 data, values in [brackets] based on ND = 0.5DL

Section 5

1. p. 5-12, Table 5-1b. “Age 12-18" should read “Age 12-19.”

2. p. 5-14, Table 5-4b. “Age 12-18" should read “Age 12-19.”

3. p. 5-15, Table 5-5b. “Age 12-18" should read “Age 12-19.”

4. p. 5-16, Table 5-6b. “Age 12-18" should read “Age 12-19.”

5. p. 5-24, Table 5-9b. “Age 12-18" should read “Age 12-19.”

6. p. 5-29, Item 4, 2nd sentence. “average” should read “median.”

Appendix C

1. Appendix C, Table C-31, Tetrachloroethylene and Table C-32, Trichloroethylene. For these two contaminants we incorrectly used cancer slope factors and unit risk factors that did not meet our selection criteria (benchmarks provided by IRIS, HEAST, NCEA, or CalEPA). These values should be deleted. There are no cancer slope factors or unit risk factors for these that meet our criteria for selecting benchmarks.

2. Appendix C, Table C-37, 2,3,7,8-TCDD. We incorrectly cited HEAST as the source of the slope factor for 2,3,7,8-TCDD. The source of this slope factor actually is:
“USEPA, 1985. Health Assessment Document for Polychlorinated Dibenzo-p-Dioxins. Office of Health and Environmental Assessment. EPA/600/8-84/014F. September.”
A risk estimate calculated using the slope factor presented in HEAST would be only a factor of 0.96 (150,000/156,000) times a risk estimate calculated based on the slope factor presented in the 1985 document. This difference would have no discernable impact on our risk estimates.

Appendix D.5

1. Section 2 of the 1999 Risk Assessment TBD said that the distance to the stream was fixed at 102 m from the source. However, stream distance actually was varied in the deterministic analyses (included as a sensitive parameter in the sensitivity analyses). The stream distances evaluated were 102 m and 300 m for the chlorinated aliphatics wastewater tank and EDC/VCM and methyl chloride landfill scenarios, and 102 m and 403 m for the EDC/VCM land treatment unit scenario. In addition, the discussion of the surface water model in Appendix D.5 of the 1999 Risk Assessment TBD suggested that we evaluated two stream orders, a high end stream of reach order 3 and a central tendency stream of reach order 5, in the sensitivity analyses for the fisher scenario. In reality, we only evaluated a stream order 3 in our fisher analysis. The rationale for selecting an order 3 stream as the fishable waterbody was based primarily on the desire to create an exposure scenario that was protective of different types of fishable waterbodies and, at the same time, representative of streams on a national basis (that is, streams of order 3 are much more prevalent on a national basis than streams of orders 4 and above).

After identifying these errors in the text, we investigated the impact of allowing the stream distance to vary and of evaluating only an order 3 stream in the deterministic analyses. Our evaluation of the risk estimates found that modifying the analysis consistent with the manner in which it is described in the 1999 Risk Assessment TBD would not result in any meaningful difference in the risk or hazard estimates. Specifically, high risk and hazard quotient estimates due to fish ingestion would be modified as follows:

- For the EDC/VCM LTU, the sum of the COPC high end risk estimates due to fish ingestion are 2E-05 and 9E-12 due to nongroundwater and groundwater pathways, respectively. The nongroundwater pathway risks are due to dioxins. The sum of the COPC high end hazard quotients due to nongroundwater pathways is 0.003 (there are no groundwater COPCs for which HQs are calculated). Modifying the analysis to evaluate a reach order 5 stream rather than a reach order 3 stream decreases the fish ingestion risk estimates by a maximum factor of 30. Evaluating a distance of 403m instead of 102m increases the fish ingestion risk estimates by a maximum factor of 1.25. This phenomenon is a function of the model construct used to simulate overland transport of contaminant through runoff and erosion of contaminant sorbed to soil particles. For constituents that sorb strongly to soil particles, decreasing the distance to receptor effectively decreases the area of the drainage subbasin, resulting in a slight decrease of the overall load due to soil erosion to the stream (see Appendix D.2 for additional details on the overland transport model). In some instances, the risk results increased with decreasing distance to stream and, in other cases, (e.g., for dioxin congeners) the risk results decreased slightly with decreasing distance to the stream.

- For the EDC/VCM landfill, the sum of the COPC high end risk estimates due to fish ingestion are 7E-15 and 1E-10, due to nongroundwater and groundwater pathways, respectively. The sum of the COPC high end hazard quotients due to fish ingestion are <0.0001 and 0.0000002, due to nongroundwater and groundwater pathways, respectively. For the nongroundwater pathways, modifying the analysis to evaluate a reach order 5 stream rather than a reach order 3 stream increases the fish ingestion risk estimates and hazard quotients by a maximum factor of 1.25. Evaluating a distance of 300m instead of 102m increases the fish ingestion risk estimates and hazard quotients by a maximum factor of 1.1. For the groundwater pathways, modifying the analysis to evaluate a reach order 5 stream rather than a reach order 3 stream decreases the fish ingestion risk estimates and hazard quotients by a maximum factor of 30. Evaluating a distance of 102m instead of 430m increases the fish ingestion risk estimates and hazard quotients by a maximum of 3 orders of magnitude.

  For the nongroundwater pathway, the increase in risk estimates/hazard quotients with a larger stream is counterintuitive because one would expect there to be greater dilution of contaminants in a larger stream. However, because we did not average the deposition rate over the entire area of the stream (a single point estimate of deposition was used for the entire surface area of the stream) we created an artificially high load of contaminant deposition to the reach order 5 stream. That is, the maximum deposition rate would have been set for the entire surface water area which, for reach order 5 streams, is over one million meters squared. Given this situation, the volumetric dilution of the contaminant would be offset by an artificially high average deposition rate such that the risk estimates for an order 5 reach could actually increase. This is particularly true for highly hydrophobic constituents because the dry vapor phase constituent diffusion load to the waterbody (L_{Dif} in equation E-4.12) is, in part, driven by the Henry's Law constant. As shown in that equation, the diffusion load tends to be higher for constituents with low values of Henry's Law constant (as with dioxins and other hydrophobic constituents). For a reach order 3 stream, the use of a single point provides a reasonable approximation of the deposition onto the stream.

- For the chlorinated aliphatic wastewater tank, the sum of the COPC high end risk estimates due to fish ingestion is 1E-10. The sum of the COPC high end hazard quotients due to fish ingestion is 0.01. Modifying the analysis to consider a reach order 5 stream rather than a reach order 3 stream increases the fish ingestion risk estimates and hazard quotients by a maximum factor of 1.5. Evaluating a distance of 300m instead of 102m increases the fish ingestion risk estimates and hazard quotients by a maximum factor of 1.2.

- For the methyl chloride landfill, the sum of the COPC high end risk and hazard quotient estimates due to fish ingestion are 4E-16 and <0.0001, respectively. These estimates would have to be modified many orders of magnitude to be of concern. Consequently, the difference in our analysis as presented in the 1999 Risk Assessment TBD and as conducted has no impact on the methyl chloride landfill fisher results.

These analyses demonstrate that modifying our evaluation of fisher receptor in a manner consistent with the methodology described in the 1999 Risk Assessment TBD would not result in any meaningful change in the risk estimates or hazard quotients calculated for the fisher.

Appendix F

1. p. F.2-7. The fish ingestion rate listed as corresponding to "all ages" should be changed to “adult.”

Appendix H

1. pp. H.3-34 and H.3-35, Tables H.3-6a and H.3-6b. The titles of these tables should indicate that they present probabilistic risk results for <u>arsenic</u>.

Appendix K

1. In Table K-3, in the column labeled *Source for Minimum and Maximum Values (If different from the Data Source)* all references to EPACMTP should be removed. All but two of these should be left blank because the source for the minimum and maximum are in fact the same as the data source. The exceptions are:

> (1) under *Saturated zone parameters*, for *Seepage Velocity (pore porosity)*, the source for minimum and maximum values should read *Minimum and maximum values based on survey data reported by Newell et al., 1990 (Groundwater 28(5), p. 703-714)*, and
>
> (2) under *Unsaturated zone parameters*, for *Saturated conductivity of the soil*, the source for minimum and maximum values should read *Minimum value set to 1.5E-04 to avoid infeasible zero value*.

2. In Table K-3, the *Bulk density* and *Porosity* parameters for the EDC/VCM landfill and the EDC/VCM LTU should be changed to read *yes* rather than *no* under the column *Monte Carlo Variable*. They currently read *no* because the values are not randomly generated from a distribution but rather are derived from data that are randomly generated. We believe that yes more accurately describes the modeling performed.

3. In Table K-4, the reference for *Beef intake* under both the *Child of Farmer (1-5 years)* and *Child of Farmer (6-11 years)* should be changed from *. . . EFH Table 13-61 . . .* to *. . . EFH Table 13-36 . . .*

4. In Table K-1, under *Tank Parameters*, the units for *Impeller diameter* should be changed from *unitless* to *cm*.

5. In table K-1, for the *EDC/VCM Wastewater Treatment Sludge Managed in an Off-site Landfill*, under *Surface and Subsurface Soil Parameters (Unsaturated Zone)* a footnote should be added to the central tendency value for *Unsaturated zone thickness (depth to water table)* which should read *Unsaturated zone thickness is a location-specific parameter. The unsaturated zone thickness values presented correspond to the central tendency and high-end locations and are varied with location, not independently.*

6. In Table K-1, for the *Methyl Chloride Wastewater Treatment Sludge Managed in an On-site Landfill,* because groundwater modeling was not performed, all checkmarks under the *EPACMTP* column should be removed. It follows that the entire row for *Average soil pH* and *Residual water content* should be deleted as the only checkmarks that were included for these parameters were for *EPACMTP*.

7. In Table K-1, for the *Methyl Chloride Wastewater Treatment Sludge Managed in an On-site Landfill* under *Landfill Parameters* the central tendency for *Type of daily cover* should be changed from *Soil* to *NA* because no daily cover is assumed.

8. In Table K-1, for the *Methyl Chloride Wastewater Treatment Sludge Managed in an On-site Landfill* under *Meteorological Parameters*, for the *Mean annual wind speed*, the checkmark under *EPACMTP* should be removed and a checkmark should be added under *ISC*.

9. In Table K-1, for *EDC/VCM Waste Managed in a Land Treatment Unit*, under *Surface and Subsurface Soil Parameters (Unsaturated Zone)* a checkmark should be added in the *Partitioning* column for the *Soil bulk density* parameter.

10. In Table K-1, under *"Wasterwaters" Managed in Aerated Tanks*, *"Wasterwaters"* should be changed to *"Wastewaters"*. All checkmarks under *Partitioning* should be removed and a checkmark should be placed under the *CHEMDAT8* column for the *Waste concentration* parameter. The units for *Waste concentration* should be changed to *mg/L*.

11. In Table K-1, under *Constants used in Fate and Transport Modeling*, under *Surface Water / Fish Parameters*, the parameters *Width of waterbody*, *Length of waterbody*, and *Total waterbody depth* should all have checkmarks in the *Indirect Exp.* column.

12. For Table K-3, all of the variables are Monte Carlo Variables, consequently the column *Monte Carlo Variable* should be deleted. Because there are no constants under the Minimum, Maximum, or Constant columns, the cell that contains *-or- Constant* should be removed.

13. For Table K-3, for *EDC/VCM Wastewater Treatment Sludge Managed in an Off-site Landfill*, under *Waste Parameters* the checkmark under *EPACMTP* should be removed for the *Contaminant concentration* parameter. Also, the checkmark under *Indirect Exp.* for the *Fraction organic carbon* parameter should be removed.

14. For Table K-3, for *EDC/VCM Wastewater Treatment Sludge Managed in an Off-site Landfill*, under *Saturated Zone Parameters*, the parameter *Groundwater Temperature* should be removed since it is not varied.

15. For Table K-3, for *Wastewaters Managed in Aerated Tanks*, under *Waste Parameters*, the checkmark under *EPACMTP* should be removed for the *Contaminant concentration* parameter and the units should be changed to *mg/L*.

16. For Table K-3, for *Wastewaters Managed in Aerated Tanks*, under *Tank Parameters*, a checkmark should be placed in the *ISC* column for the *Surface area* parameter.

17. For Table K-3, for *Wastewaters Managed in Aerated Tanks*, under *Tank Parameters*, *Site-specific* should be entered in the *Variable Type* column for *Soil saturated volumetric water content*, *Saturated hydraulic conductivity*, and *Soil specific exponent representing water retention.*